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Information Problems in the Design of Nonpoint Source Pollution Policy*

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Information Problems in the Design of

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I. Introduction

Large industrial and municipal emissions were the focus of first-generation environmental policies of the 1970's. Twenty years later, with much success in cleaning up industrial and municipal sources, the focus has changed. The problems of the moment include hazardous wastes, solid wastes, auto pollution, nutrient pollution, pesticide pollution, and sedimentation. These problems, by and large, are caused by many small polluters--such as users of weed sprays, motorists, farmers, and generators of household trash.

A common denominator of the contemporary problems is a high degree of difficulty in keeping track of individual pollution sources. There are so many sources that monitoring all of them would be prohibitively expensive. Furthermore, in many cases, the pollution is not a distinctive discharge, but rather is a diffuse side effect of complex activities (e.g., farming operations). This also hampers monitoring at the source. In addition, it is generally difficult to infer the pollution originating from any individual source from observations of ambient pollution levels, since ambient levels are determined by the combined activities of many polluters as well as random factors over which polluters have no control. The second generation pollutants also have complex environmental fates. For example, dissolved fertilizers break down into several chemical forms as they move into surface and ground waters. Dislodged soil particles move around in space creating flooding, degrading fish habitat, and increasing water treatment costs. With the potential to affect several media, these pollutants may be the focus of multiple policy objectives. The appropriate policy tactics may change over space and time.

We have, then, a set of complicated environmental problems that do not fit neatly into the first generation mold. Rather, the second generation problems involve polluters who are difficult to identify, emissions that are virtually impossible to monitor, and environmental fates that are multifaceted and uncertain. Information problems are at the root-of all of these difficulties.

These information problems greatly complicate the selection and implementation of policies to control second generation pollutants. The common and most direct prescription for controlling pollution, namely taxing or regulating emissions, is not a viable option for controlling these second generation pollutants. Instead, indirect policies applied to something other than emissions must be used. Examples include up-line policies (such as taxes or regulation) applied to input use and down-line policies (taxes or liability) based on ambient pollution levels. The question is then whether these indirect instruments can serve as perfect substitutes for direct control of emissions.

In general, these indirect policies are likely to be imperfect substitutes for direct emissions control. Some of the same information problems that prevent the use of direct emission control policies also imply imperfections in using indirect policies. For example, pollution-related inputs that are not easily monitored are not amenable to taxation or regulation. Likewise, the use of ambient taxes or liability can be hampered by possible information problems such as identifying the actual or probable contribution of individual sources.

If no single direct or indirect policy instrument can ensure efficient abatement of these second generation pollutants, then what recourse do environmental economists and policy makers have? Clearly, one approach is simply to live with the imperfections and analyze individual policies in a second-best context. While this seems to be a common approach of economists,[1] policy makers seem to have chosen an alternative approach. Rather than searching for the “best” (in a second-best sense) single instrument, policy makers appear instead to be searching for a combination of indirect instruments to control these second generation pollutants. In a world of first-best instruments, simultaneous use of instruments is at best redundant and at worst counter-productive. However, when information problems prevent the use of first-best instruments, a theoretical rationale for combining instruments may exist. In searching for a combination of instruments, policy makers may in fact be ahead of the theory of efficient pollution control, which has focused almost exclusively on single-instrument approaches.

Examples of multiple instrument approaches are easy to find. As illustrations: A host of initiatives, from litter laws and beverage container deposits to recycling programs and mandated use of recyled paper by government agencies, are aimed at reducing the disposal of solid wastes. The prevention of pesticide contamination is the objective of complex licensing and labelling standards, food safety standards, and, potentially, products liability. And, the abatement of sedimentation is promoted through erosion control standards, government subsidies for erosion control, and, potentially, nuisance or tort law remedies.

In this paper, we consider the choice of environmental policies under incomplete information, with special reference to nonpoint source pollution (NSP) and the types of policy instruments that could be used in this context. In considering nonpoint source pollution, we will focus particularly on pollution from agricultural land uses. [2] Agricultural NSP is widespread and of current concern in many countries. It certainly is prone to many of the challenges noted above: the many, dispersed sources are difficult to identify; monitoring is nearly impossible because of the diffuse sources; several media are affected; and the occurance and impacts of agricultural NSP are nearly impossible to predict because of the importance of stochastic weather and production variables.

We begin by examining in greater detail the particular information problems of agricultural NSP and explore some of the implications for policy design. Next, we focus on the simultaneous use of multiple instruments as a means of compensating for information problems. Using a very simple model, we show that multiple instruments can be an efficient response to imperfections in single instruments due to information problems, although care must be taken that the instruments that are combined are complementary rather than contradictory.

II. Information Problems with Nonpoint Source Pollution

Figure 1 represents a general relationship between sources and ultimate impacts of pollution for two firms whose emissions combine to determine ambient environmental quality at a particular location. The figure depicts the various steps in the production process of a firm, from its initial input/technology choice to its products, emissions, and ultimate environmental or health impacts. The emissions, perhaps of several types and from multiple sources, may affect several environmental media, sometimes interconnected media (such as ground and surface water often are), and cause contamination. Exposure of susceptible humans, other life forms, or physical systems to the contamination leads to damages being incurred.

The description in Figure 1 seems general enough to fit pollution from both point sources and nonpoint sources. Of interest to us are the informational characteristics of the various steps of the pollution process and the specific information problems of agricultural nonpoint source pollution. We will discuss two classes of information problems: natural variability and problems of monitoring and measurement.

Natural Variability

As indicated symbolically in Figure 1, pollution processes are affected by various natural sources of variability, including weather, mechanical malfunctions, and susceptibility to damages[3]. As a result, a particular policy (or a specific abatement plan) will produce a distribution of outcomes rather than single outcome.

If the outcomes cannot be precisely foreseen, then abatement policies and abatement methods must be evaluated according to their effects on the expected distribution of outcomes, as determined by the distributions of the underlying random variables. This randomness does not, by itself, prevent the attainment of *ex ante* efficiency through the use of standard policies.[4] In other words, if neither the firm nor the social planner knows the values of the random variables at the time that decisions are made, and if both are risk neutral, then the policy maker can structure a tax or regulation that will cause rational private decision makers to act in a way that maximizes the expected value of social surplus. For example, the planner can place a tax on polluting inputs equal to the expected marginal external cost of using the input, thereby ensuring that expected marginal social costs equal expected marginal private costs. In such a case, randomness should not affect the selection of a policy goal, although realization of that goal at any one time will be a random event. However, randomness can affect the relative *ex post* efficiency of different policy instruments, as shown by Weitzman (1974).[5]

Another type of variability has to do with space rather than time. A state or national environmental policy must apply in a variety of local circumstances as well an enduring variation over time. As shown by Kolstad (1987), certain policy instruments may contend with diverse local circumstances more efficiently, in an *ex post* sense, than others. Like Weitzman's (1974) analysis, the relative curvatures of abatement cost and benefit functions determine whether incentive or regulatory instruments are more robust when applied acres local circumstances.

The expected mean value of emissions or ambient contamination in many cases is a sufficient *ex ante* measure of an environmental goal.[6] However, in some instances, deviations around the mean are important as well. For example, variation in pollution outcomes is often incorporated into regulatory policies by setting a threshold level of environmental quality (Q*) and a safety margin, expressed as a maximum acceptable frequency (1-P) of exceeding the threshold (Lichtenberg and Zilberman, 1988; Braden, Larson, and Herricks, 1991):

$$\text{Prob}\,(Q \geq Q^*) \leq 1 - P\ ,$$

where Q is measured environmental contamination and P is the cumulative probability of Q. Such a policy goal calls for abatement measures that will not only affect the mean realizations of abatement (keyed to the threshold), but also the variability of the pollution distribution (in reaction to the safety margin). Unless the mean realization and the variance are correlated, a single policy instrument will not generally achieve the joint goal in an efficient manner. Combining instruments that apply to specific moments of the distribution will often enhance efficiency.[7] For example, in addition to specifying maximum customary rates, emissions regulations frequently specify special rates that apply when background conditions are less able than usual to assimilate pollutants.

To summarize the preceding discussion: Even in the presence of natural variability, policy instruments can be selected to achieve *ex ante* efficiency, although the resulting level of environmental quality will deviate from the *ex post* socially efficient goal. A similar conclusion applies when a single policy must address a problem that varies from place to place. In addition, if damages are affected by higher moments of the distribution of ambient quality, then the use of several policy instruments may enhance efficiency under some circumstances. The lessons for agricultural NSP policies depend on the particular empirical properties of the abatement supply and demand curves, on the spatial variation in the problems, and on the importance of and relationships between moments of the distribution of outcomes.

Empirical research on agricultural nonpoint source pollution has benefitted from simulation models of pollution processes[8] that provide insight into the costs of abating agricultural NSP.[9] Unfortunately, there is virtually no corresponding information on benefits (abatement demand).[10] The cost studies indicate that, at least for sediment, the supply curve begins with very little slope and becomes steeper as abatement goals are raised. Illustrative abatement supply curves for sediment, taken from a study of Central Illinois conditions by Braden *et al.* (1989), are reproduced in Figure 2.[11]

With little information on abatement demand, we can only speculate about the ranking of incentive and regulatory policies. If demand and supply intersect at low levels of abatement, then the demand curve would almost certainly be steeper than the very flat supply curve and an abatement standard set to achieve the expected pollution level would probably minimize the *ex post* losses in economic surplus. At the other extreme, the steep portion of the cost function would almost certainly be steeper than an intersecting demand curve, in which case an incentive instrument would minimize the *ex post* losses.[12]

On the matter of spatial variation, at least with respect to abatement costs, the empirical literature provides more to go on. Park and Shabman (1982) analyze the value of regional "targeting" (differentiated policies) while Braden *et al.* (1989) analyze the value of micro-targeting within a watershed. Both indicate that spatially uniform policies are inefficient. The finding of significant *ex post* inefficiency underscores the merit of locally differentiated or flexible policies rather than uniform policies. However, as between uniform taxes and regulations applied to simulated erosion rates, Miltz, Braden, and Johnson (1988) suggest that it may not be possible to draw general conclusions about which is more efficient. Their results indicate that taxes achieve modest reductions in simulated sedimentation at a lower cost while regulations achieve extreme reductions at a lower cost.

Finally, higher moments of the distribution of outcomes are environmentally important for several agricultural pollutants. For example, extreme concentrations of some agricultural chemicals can be acutely toxic while average concentrations have no effect. Sediment is also illustrative--average loads are relevant to depletion of reservoir storage capacity while extreme loads play a major role in flood damages. As noted above, several instruments maybe needed to abate most efficiently the multifaceted damages.

Imperfect Monitoring and Measurement

In addition to natural variability, various aspects of pollution production processes are subject to imperfect monitoring and measurement. Many elements cannot be easily monitored. Others are likely to be monitored only occasionally, so unusual occurances may go undetected. In addition, malfunctioning or insufficiently sensitive testing equipment can provide misleading information.

With imperfect monitoring and measurement, policy enforcement will also be imperfect (Russell, Harrington, and Vaughan 1986). Violations may not be detected ("false negatives") or may be spuriously inferred ("false positives"). The social costs of these errors are of two types: 1) the damages (net of abatement costs) that would have been prevented if violators could have been induced to comply with policies, and 2) the excessive abatement costs (net of abatement benefits) resulting from unfounded enforcement actions. These potential costs must be weighed against the costs of more complete testing and more precise measurement. Vaughan and Russell (1983) illustrate the use of statistical quality control measures to devise an optimal monitoring regime.

Imperfections in monitoring and enforcement are not only potentially costly, they also create opportunities for polluters to influence the information that becomes available to enforcement officials. For example, the enforcement of many pollution control laws is based, in the first instance, on self-monitoring data reported by polluters. These reports are periodically verified through pre-announced site visits by government officials. If it wished, the polluting firm could falsify its reports and misrepresent the typical plant operations during the periodic site visits.

The potential for cheating makes measurement error, in part, an endogenous consequence of the choice of abatement policies. The incentives to cheat can be influenced through more intensive and/or less predictable monitoring and through penalties for misrepresentation as well as for violations (Polinsky and Shaven 1979).[13]

Cheating is one manifestation of a more general problem--information asymmetry. Information on actual production practices, emissions, and costs is available to a polluting firm but often unavailable to a regulatory agency. Information asymmetries can take two basic forms: moral hazard (inability to observe inputs) and adverse selection (inability to observe technology or type). A number of studies have examined the implications of asymmetric information regarding pollution control[14].

In the context of agricultural NSP, information problems related to imperfect monitoring arises in at least three ways: (1) the inability to observe emissions, (2) the inability to infer emissions from observable inputs, and (3) the inability to infer emissions from ambient environmental quality. While no one of these by itself necessarily prevents the design of an efficient pollution control instrument, the combination of the three makes policy design in this context particularly challenging.

Unobservability of Emissions. The inability to observe emissions is the single most troublesome characteristic of nonpoint source pollution and the feature that most distinguishes NSP from point source pollution.[15] Monitoring of NSP emissions is impractical, since emissions are by definition diffuse. For example, measuring the amount of soil lost from a particular field or the amount of a chemical leaving the root zone en route to a nearby aquifer would require monitoring over the entire field rather than at a single location in the field. The associated monitoring costs are prohibitively expensive.

The inability to observe emissions impedes the use of the single most common environmental policy instruments--the emission standard. The lack of observability also undercuts the use of emission taxes, complicates the application of liability (Miceli and Segerson 1991), and diminishes accountability for abatement incentives.

Of course, the inability to observe emissions could be circumvented if the level of emissions were perfectly correlated with some other observable part of the production or pollution process, such as an input or ambient quality (Nichols 1984). In this case, a tax or standard on the input or the ambient quality could serve as a perfect substitute for an emissions tax. However, as discussed below, such a close correlation is unlikely. In the absence of close correlation, a policy based on a particular input or ambient condition could diminish efficiency by biasing the selection of inputs or failing to account for differences in emissions.[16]

Unobservable Inputs/Technology Many agricultural nonpoint source pollutants are closely associated with specific, readily observable production inputs. For example, pesticide contamination is closely associated with pesticide use; more particularly, it is associated with the pesticides that are applied to specific crops grown in porous soils over shallow aquifers. The amounts of pesticides purchased, the crops being grown, and the physical circumstances can all be determined by a regulatory agency. Similarly, erosion is closely associated with certain crops, soils, and tillage techniques, and these are readily inspected.

However, agricultural pollution levels are likely to depend not only on these observable inputs, but also on some critical, unobservable inputs. For example, the pollution resulting from a given quantity of pesticide applied may depend not only on the total quantity applied but also on the care with which it is prepared, the timing of application, and where it is applied (such as how close to streambanks or wellheads). While these timing and application inputs are theoretically observable, observations by a regulatory body would require continual monitoring of farm operations, which is impractical.

The unobservability of some key inputs implies that these inputs cannot be subject to direct control through regulations or taxation. In addition, taxing or regulating only the observable inputs will generally distort the chosen input mix and induce inappropriate substitutions.

The inability to control inputs directly is a classic moral hazard problem. The usual prescription is an output-based incentive instrument. With agricultural NSP, such an instrument would have to be based on ambient environmental quality. As we discuss below, this is not an entirely satisfactory solution, since information problems are likely to hamper the efficiency of such policies. Fortunately, however, in some respects, a farmer's personal economic interest may deter environmentally egregious uses of inputs, such as wasteful chemical applications.[17] To the extent that private costs and benefits cause farmers to use timing and application methods that reduce runoff and leaching in order to increase efficacy, the moral hazard problem from unobservability of these inputs is reduced.

Inferring Emissions from Ambient Pollution. Since ambient pollution levels are relatively easy to observe, they can provide information about the extent of polluting activities in the vicinity of a given environmental medium. Unfortunately, however, while it may be relatively easy to observe contamination levels (such as the turbidity of a stream or the level of contamination of an aquifer), attributing that contamination to a given level of emissions at a particular source may be very difficult. For example, determining the origin of particles deposited in a stream is virtually impossible.

The inability to infer emissions from observed ambient pollution is the result of both natural randomness and the influence of other neighboring polluters. If many polluters border a particular stream or overlie a particular aquifer, then the level of contamination is determined by their combined activities. In addition, the effectiveness of abatement measures undertaken by one firm depends on the actions taken by others.

Despite the inability to attribute a given level of ambient pollution to the activities of individual polluters, Segerson (1988) and Xepapadeas (1991) have shown that, at least in theory, an ambient tax/subsidy scheme can provide the correct incentives for individual polluters to undertake socially efficient abatement measures. [18] Under the proposed policy, each polluter (actual or potential) would be required to pay an ambient-based pollution tax (or receive a subsidy) equal to the full marginal social cost (benefit) of the collective level of contamination (abatement).[19] Even with multiple polluters, this approach provides each polluter with the socially efficient marginal incentive to abate.[20] Polluters for whom management changes will have little impact on contamination will have less incentive to abate than those whose management changes will have a large effect. The tax would also encourage the most efficient means of abatement, be it reducing inputs or modifying technology.

While in theory the above proposal ensures first-best incentives even in the presence of multiple polluters, it suffers from several practical difficulties. For example, setting each polluter's efficient tax rate requires extensive information on the entire process outlined in Figure 1 for each polluter contributing to the contamination. [21] This presents a serious information burden and maybe impractical, Furthermore, each polluter's tax exposure depends in part on the pollution of others. A uniform tax could be criticized as equal punishment for unequal pollution.

Practical difficulties in monitoring ambient quality may reduce the incentive effects and, hence, the efficiency of an instrument applied to ambient contamination. Ideally, ambient-based taxes would be implemented on the basis of continuous monitoring of environmental quality. The policy signals sent to polluters then could be continually adjusted according to actual circumstances. However, this ideal is far from realistic. A more likely scenario is the periodic taking of samples in a sparse network of monitoring sites. The policy signals would be based on extrapolations to unmonitored sites and times. In such a setting, abatement efforts will have only a tenuous effect on the measured outcomes. Accordingly, polluters will be discouraged from undertaking socially desirable abatement.

An alternative to ambient-based incentives is *ex post* liability for contamination or damages. This approach provides a potential solution where only some unpredictable subset of all emissions cause damages and the transactions costs are modest for seeking compensation for individual episodes.

Liability works only when causality can be established. Thus, information must be available to establish the reality of harm and the responsibility for having caused it. The inability to observe emissions, coupled with the inability to attribute ambient pollution to any individual farm due to natural randomness and the influences of multiple polluters, implies that causality may be difficult to establish or prove in many cases of nonpoint source pollution.[22] As such, even if polluters are theoretically liable for damages under either statute or the law of torts, there is a significant positive probability that they would not actually be held liable. This clearly reduces the incentives for pollution abatement.

Another practical difficulty with liability remedies is that the expected liability for damages as viewed by tortfeasors is likely to be below the expected value of damages The difference is due to the potential to avoid damage claims through bankruptcy, the less than certain likelihood of suits by victims, and the possibility of an inappropriate verdict (Shaven 1984 and Kolstad, Ulen, and Johnson 1990). In the case of agricultural NSP, all three factors seem pertinent, but especially the uncertainty about an appropriate verdict (since farmer liability for environmental damages is only now beginning to be considered) and bankruptcy (since most farms are small enterprises with limited capacity to spread the risk of a damage claim). Under these circumstances, liability alone cannot be counted upon to balance social costs and benefits.

There are certain types of problems, however, for which liability might be effective. One is the case of manufacturer liability for damages due to pesticide contamination of groundwater. Here, bankruptcy is less of a problem since many chemicals are produced by large companies. In addition, liability for damages from products is a well-established field within tort law, suggesting that the legal system has established mechanisms for dealing with such cases. Finally, for many chemicals, a distinctive chemical “fingerprint” removes doubt about the “responsible party”, in terms of the manufacturer. Segerson (1990) establishes that producer liability has consequences equivalent to perfect application of user liability, in that producers will increase the prices of pesticides to fund their expected liability exposure. Thus, holding the manufacturer strictly liable has the same effect as charging the chemical user for damages. The liability will cause the manufacturer to assess the financial exposure and raise the chemical price accordingly. The assessment, and the resulting price increases or users warnings, may even take into account different levels of risk in different physical settings--for example, where soils are more permeable or ground water resources are closer to the surface. Such price increases would discourage use of the chemical just as taxes would. However, if contamination is affected by timing and method of use, manufacturer liability alone may not ensure that these dimensions are efficiently exploited.

III. Multiple Instruments as a Response to Information Problems

With the information problems discussed above and the many facets of agricultural nonpoint source pollution, no single policy instrument is likely to yield efficient pollution abatement decisions. Input taxes applied only to observable inputs will ignore the role of unobservable inputs, thereby distorting input choices. Likewise, while the use of ambient-based policies avoids the need to control input use directly, it is likely to lead to an imperfect internalization of costs due primarily to the inability to attribute ambient pollution to the activities of individual polluters.

Rather than frame the problem as a choice between two imperfect approaches, we suggest that a preferred approach may be to combine policy tools into a policy “package.” While we have made similar suggestions previously (Braden 1990 and Segerson 1990a), we are unaware of any formal analyses of the welfare effects of a multiple instrument approach in the presence of information problems. The use of multiple tools or instruments is redundant in a world of first-best single instruments, but it may have a role to play in improving efficiency when single instruments are imperfect.[23]

In this section we consider a very simple model that illustrates the role that multiple instruments can play in the control of nonpoint pollution. For simplicity, we consider only two information problems: (1) the inability to observe (and thus tax) all pollution-related inputs, and (2) the chance that a responsible party may not be held liable for damages under liability due to difficulties in identifying the source and establishing causation.[24] We show that, while the sole use of an input tax (on the observable input) or liability will not be efficient, combining the two policies may improve social welfare. This result is not guaranteed, however, since in some cases combining policies can actually reduce welfare. The result depends upon the way in which pollution-related inputs interact with each other in both the production and the pollution process. This suggests the need for care in combining policies to ensure complementarity between the individual policies.

Consider a farm that uses two inputs, X and Y, to produce an output. Let the net private benefits from the production process be NB(X,Y), with $NB_x > 0$ and $NB_y > 0$. (Subscripts on functions denote partial derivatives.) NB is assumed to be strictly concave in (X,Y), implying $NB_{xx} \leq 0$, $NB_{yy} \leq 0$, and $NB_{xx}NB_{yy}-(NB_{xy})^2 \geq 0$.

Use of the inputs is also assumed to result in an expected level of damages from ambient pollution, denoted D(X,Y). To the extent that damages are influenced by random variables such as weather, D will depend on both the probability distributions of these random variables and the set of possible outcomes. For simplicity of notation and without loss of generality (given risk neutrality), we subsume these random effects into the D function, which represents expected damages. In addition, if there are multiple polluters, expected damages may also depend on the actions of other farms. In this case, D would have additional arguments reflecting the decisions of other firms. We do not consider the role of other firms explicitly, since doing so would complicate the exposition without changing the basic qualitative conclusions. Finally, damages could result from contamination of several environmental media. For example, D could represent combined impacts on groundwater and surface water (i.e., $D=D^s+D^g$, where D^i denotes damages to media i, with i=s (surface water) or g (groundwater)). We assume that $D_x>0$ and $D_y>0$, i.e., that increases in either of the inputs would increase aggregate damages. This does not imply, however, that tradeoffs between different media do not exist. For example, increases in input X may increase groundwater contamination ($D^g_x>0$) while decreasing surface water contamination ($D^s_x< 0$). We simply assume that, on net, the effect is an increase in overall damages. Finally, we assume that damages are convex in (X, Y), i.e. $D_{xx}\geq 0$, $D_{yy}\geq 0$, and $D_{xx}D_{yy}-(D_{xy})^2\geq 0$.

Expected social net benefits from the farm's production process are SNB(X,Y)=NB(X,Y)-D(X,Y). The first-order conditions for the maximization of expected social net benefits are:

(1) $NB_x - D_x = 0$, and

(2) $NB_y - D_y = 0$.

Given the curvature assumptions on NB and D, equation (1) defines the efficient level of X given Y, which we denote X*(Y). Likewise, (2) defines the efficient level of Y given X, Y*(X). Simultaneously, (1) and (2) define the efficient levels of X and Y, (X^*_e, Y^*_e).

We consider three alternative policy approaches that could be used to internalize the farm's external pollution costs. The first approach is the use of input taxes. However, because of information problems, we assume that not all inputs can be monitored and thus subject to direct taxation. In particular, we assume that, while the regulatory agency can easily observe (and thus tax) the X input, it is unable to tax the Y input. Thus, the first policy alternative is simply to impose a per-unit tax on X, with the level of the tax equal to the marginal external damages from use of X, i.e., $t=D_x$.. Faced with such a tax, the farmer would choose the levels of X and Y to maximize NB(X,Y)-tX, yielding the following first-order conditions:

(3) $\quad NB_x - D_x = 0$, and

(4) $\quad NB_y = 0.$

Note that (3) is identical to (1). Thus, under the input tax approach, the firm would choose the efficient level of X given Y. However, since (4) differs from (2), it would not choose the efficient amount of Y given X. Let $Y_o(X)$ denote the solution to (4) given X. $Y_o(X)$ and $X^*(Y)$ simultaneously determine X^*_t, Y^*_t), the input choices under the input tax approach, which will be inefficient.

The second policy approach is to use instead an ambient-based policy such as liability for actual damages. Under this approach, if held responsible for contamination, the farmer would expect to pay an amount equal to the resulting damages. However, again because of information problems, there is some probability that parties responsible for pollution will not be easily identified and thus held liable for the associated damages. Let $p<1$ be the probability that the firm will actually have to pay for the expected damages that it creates.[25] Then, under the liability policy, the firm would choose X and Y to maximize NB(X,Y)-pD(X,Y), yielding the following first-order conditions:

(5) $\quad NB_x - pD_x = 0$, and

(6) $\quad NB_y - pD_y = 0.$

Let $X_L(Y)$ denote the solution to (5) given Y, and let $Y_L(X)$ denote the solution to (6) given X. The simultaneous solution of the two equations gives the input choices under the ambient-based policy (X^*_L, Y^*_L). Note that in this case neither X nor Y is chosen efficiently, given the level of the other input.

Finally, policy makers can use a multiple-instrument approach, under which they combine the use of an input tax on X and liability. If policy makers recognize the imperfections in the use of the liability policy, they can add an input tax on X to try to completely internalize the external costs resulting from the use of X. Alternatively, if they recognize that the external costs from using Y are not internalized through the input tax approach, they can add, for example, a liability rule to try to influence indirectly the choice of Y. It should be noted, however, that when the two policies are combined the level of the input tax that will fully internalize the costs of X will no longer equal marginal expected damages, D_x. Since the marginal effect of liability will impose costs of pD_x, the input tax should simply reflect the remaining costs that have not been internalized, i.e., $(1-p)D_x$.

Under this combined approach, the firm will choose X and Y to maximize NB(X,Y)-pD(X,Y)-tX, where $t=(1-p)D_x$ evaluated at the efficient level of X given Y. This yields the following first-order conditions:

(7) $$NB_x - pD_x - (1-p)D_x = NB_x - D_x = 0, \text{ and}$$

(8) $$NB_y - pD_y = 0.$$

Note that, since (7) and (1) are identical, again the firm chooses the efficient level of X given Y. In addition, comparing (8) and (6) implies that it chooses the same level of Y given X that it would have chosen had a liability rule been used alone. However, the combined solution to (7) and (8), (X^*_c, Y^*_c), will in general differ from the input choices when either of the two policies is used alone.

Our objective is to compare expected social net benefits under the single-instrument approaches (input tax alone or liability alone) to the expected social net benefits when the instruments are combined, to determine if the use of multiple instruments improves social welfare.

Consider first the comparison of the tax alone to the tax coupled with liability. Under the tax alone, expected social net benefit is given by $SNB(X^*_t, Y^*_t)$. Likewise, expected social net benefit under the combined approach is $SNB(X^*_c, Y^*_c)$. Note, however, that

$$\text{(9)} \qquad SNB(X^*_t, Y^*_t) = SNB(X^*(Y^*_t), Y^*_t) = S\tilde{N}B(Y^*_t), \text{ and}$$

$$\text{(10)} \qquad SNB(X^*_c, Y^*_c) = SNB(X^*(Y^*_c), Y^*_c) = S\tilde{N}B(Y^*_c).$$

Thus, to compare the two approaches, we need simply to determine whether $S\tilde{N}B(Y^*_t)$ is greater or less than $S\tilde{N}B(Y^*_c)$.

It can be easily shown that Y*, > Y*C > Y*=, i.e., that imposing liability (in addition to the tax on X) will decrease the use of Y, but with p< 1 the resulting use of Y will still exceed the efficient level. Furthermore, by definition of $S\tilde{N}B$ and Y^*_e, Y^*_e maximizes $S\tilde{N}B(Y)$. Thus, as illustrated in Figure 3, it must be true that $S\tilde{N}B(Y^*_c) > S\tilde{N}B(Y^*_t)$. In other words, combined use of liability and a tax on X must result in a higher level of social welfare than use of the tax on X alone.

The intuition behind this result is straightforward. With the input tax alone, the tax can be set to ensure the efficient level of X <u>given Y</u>, although it does not ensure the efficient level of Y. Thus, there is only one distortion in the firm's production decision, namely, the distorted choice of Y (too much Y, given X). Adding liability reduces the level of Y, thereby reducing the distortion and improving social welfare.

Unfortunately, the conclusions are not so straightforward when the combined approach is compared to the use of liability alone. In particular, we show next that use of liability alone can in some cases yield higher social welfare than the combined use of liability and an input tax.

When liability is used alone, expected social net benefit is given by

$$(11) \qquad SNB(X^*_L, Y^*_L) = SNB(X^*_L, Y_L(X^*_L)) = \hat{SNB}(X^*_L).$$

Likewise, expected social net benefit under the combined approach is

$$(12) \qquad SNB(X^*_c, Y^*_c) = SNB(X^*_c, Y_L(X^*_c)) = \hat{SNB}(X^*_c).$$

The desirability of the two approaches then depends on whether $\hat{SNB}(X^*_L)$ is greater or less than $\hat{SNB}(X^*_c)$.

As before, we can easily show that $X^*_c < X^*_L$, i.e., that adding the tax on X to a pre-existing liability rule will decrease the use of X since it increases the firm's marginal cost of X. To rank $\hat{SNB}(X^*_L)$ and $\hat{SNB}(X^*_c)$, we then need to determine if $\hat{SNB}(X)$ is increasing or decreasing at X^*_L and X*C. The slope of $\hat{SNB}(X)$ is given by

$$(14) \qquad \hat{SNB}_x = (NB_x - D_x) + [NB_y - D_y](dY_L/dX).$$

However, using (6) (which defines $Y_L(X)$), this can be re-written as

$$(15) \qquad \hat{SNB}_x = (NB_x - D_x) - (1-p)D_y(dY_L/dX).$$

At X^*_c, $NB_x - D_x = 0$ by (7). Likewise, by (5), $NB_x - D_x < 0$ at X^*_L. Thus, to determine the sign of $\hat{SNB}_x$, we need to determine the sign of dY_L/dX.

The definition of $Y_L(X)$ (equation (6)) implies that

(16) $dY_{L'}/dX = -[NB_{xy} - pD_{xy}] / [NB_{yy} - pD_{yy}]$.

From the curvature assumptions, the denominator of (16) is negative. However, the sign of the numerator depends on the interactions between the two inputs both in the production process and in determining ambient pollution. Without empirical information on these interaction effects, the sign of $dY_{L'}/dX$ cannot be determined. We thus consider the two possible cases ($dY_L/dX > 0$ and $dY_L/dX < 0$) separately.

Suppose $dY_L/dX > 0$. In this case, from (15) it is clear that $\hat{SNB}_x < 0$ at both X^*_c and X^*_L. Thus, since $X^*_c < X^*_L$, it must be true that $\hat{SNB}(X^*_c) > \hat{SNB}(X^*_L)$. (See Figure 4.) In other words, if $dY_{L'}/dX > 0$, then the combined use of liability and an input tax will be preferred to the use of liability alone.

Suppose instead that $dY_L/dX < 0$. In this case, $SNB_x > 0$ at X^*_c, while SNB_x can be positive or negative at X^*_L. If $SNB_x > 0$ at X^*_L, then again we can unambiguously rank the policy alternatives (see Figure 5). However, in this case, the use of liability alone is preferred to the combined policy. In other words, imposing an input tax on X on top of a pre-existing liability rule will unambiguously decrease social welfare. This illustrates a case where the use of multiple instruments would actually be counterproductive.

Alternatively, if $dY_L/dX < 0$ but $SNB_x < 0$ at X^*_L, then the results are ambiguous. Depending on the levels of X^*_L and X^*_c, it is possible for liability alone to be preferred (see X^*_L' in Figure 6) or for the combined approach to be preferred (see X^*_L" in Figure 6). Thus, the two policy approaches cannot be unambiguously ranked in this case.

The ambiguity that arises when comparing the use of liability alone to the use of liability plus a tax on X can be explained as follows. When liability alone is used and enforcement is imperfect, there are two distortions present. Neither X nor Y is efficient, given the level of the other input. Adding a tax on X (appropriately set to reflect the existing liability rule) will eliminate the distortion on X, thereby reducing the number of distortions to one. However, it is well-known from the theory of the second best that eliminating one distortion in a world of multiple distortions will not always improve welfare. In this case, the effect on welfare depends on how the tax affects the choice of Y (through the effect on X). How Y responds will depend on the nature of the synergisms (both in production and pollution) between X and Y. If Y stayed constant (at its level with liability alone) or decreased and X decreased when the tax was imposed, then the result would be an unambiguous increase in social welfare. However, in general a change in X could result in an increase in Y. If Y decreases also, then the effect on Y reinforces the effect on X and welfare unambiguously increases. However, if Y increases in response to the decrease in X, then taxing X would actually exascerbate the distortion in the choice of Y. If this effect is sufficiently large to offset the gain from the reduction in X, then welfare could actually decrease as a result of imposing the tax.[26]

The above analysis suggests that the ability of a multiple-instrument approach to combat effectively the information problems inherent in nonpoint source pollution hinges on both the nature of the single instruments that are considered and the nature of the interactions between pollution-related inputs. Thus, designing an effective policy package requires empirical information on these interactions. Unfortunately, little attention has been focused on this issue to date.

IV. *Information Generation as a Direct Policy Objective*

We have argued that information problems prevent the standard policy tools from individually achieving efficient incentives to control nonpoint pollution and that combining instruments may improve efficiency. At the same time, better information about pollution processes could increase the prospects for efficiency in the choice of policy goals and instruments through better ability to forecast the outcomes of particular policy choices. Measures to improve information may be thought of as distinct policy instruments which may be part of a multiple instrument approach to abatement.

For examples: Data provided by the quintennial Natural Resource Inventories, which were initiated in the late 1970's by the federal Soil and Water Resource Conservation Act, have improved the capacity to direct erosion control subsidies toward environmental needs. Data generated in recent years by the U .S. Environmental Protection Agency's national ground water survey have clarified the extent and nature of contamination problems. Data on chemical use will be enhanced by the provisions of the 1990 Farm Bill, that require participants in various farm subsidy programs to keep detailed records on chemical use. In addition, pesticide registration and licensing regulations typically require the keeping of detailed records on applications of restricted use chemicals. These data may contribute in future years to a better understanding of the factors causing pesticide contamination. Similarly, many states now require extensive monitoring of groundwater quality, particulary around landfills and hazardous waste sites, which will provide early warning of problems as well as data for use in understanding causes.

These and other information discovery policies have economic value insofar as they improve the efficiency with which other policy instruments can be applied. Perhaps the most compelling need is for data that will support the selective application of policies to areas that have or to pollutants that cause especially serious problems. Furthermore, through a more complete understanding of the connections between the sources and fates of contaminants, additional information may make *ex post* liability more compelling and more efficient as a mechanism for promoting abatement of nonpoint source pollution.

V. Conclusions

With nonpoint source pollution, natural variability in pollution processes and imperfections in monitoring and measurement (enforcement) complicate the design and implementation of policies. The single most troublesome information problem is the inability to observe NSP emissions. Without this information, pollution control policies must be indirectly applied, through regulations or incentives on input use or ambient conditions. But, indirect instruments are themselves subject to information problems which limit their capacity to achieve an efficient solution. We have shown above that the use of multiple indirect instruments may promote efficiency where single instruments cannot because of information problems.

Our analysis pertains to the stylized facts of public policies toward a variety of nonpoint source pollutions, especially agricultural NSP. Those facts include the simultaneous use of multiple instruments, such as standards plus cost-sharing for erosion control and input regulations plus liability for pesticides. However, the reinforcing effects of multiple instruments are not guaranteed. An indirect policy applied to one component may push firms toward production systems that are more polluting rather than less-polluting. Greater insight into these interrelationships could enhance the efficiency of a multiple instrument approach. Thus, generating better information can bean important foundation for a pollution control strategy that also includes other instruments.

Figure 1. Information about Pollution Relationships

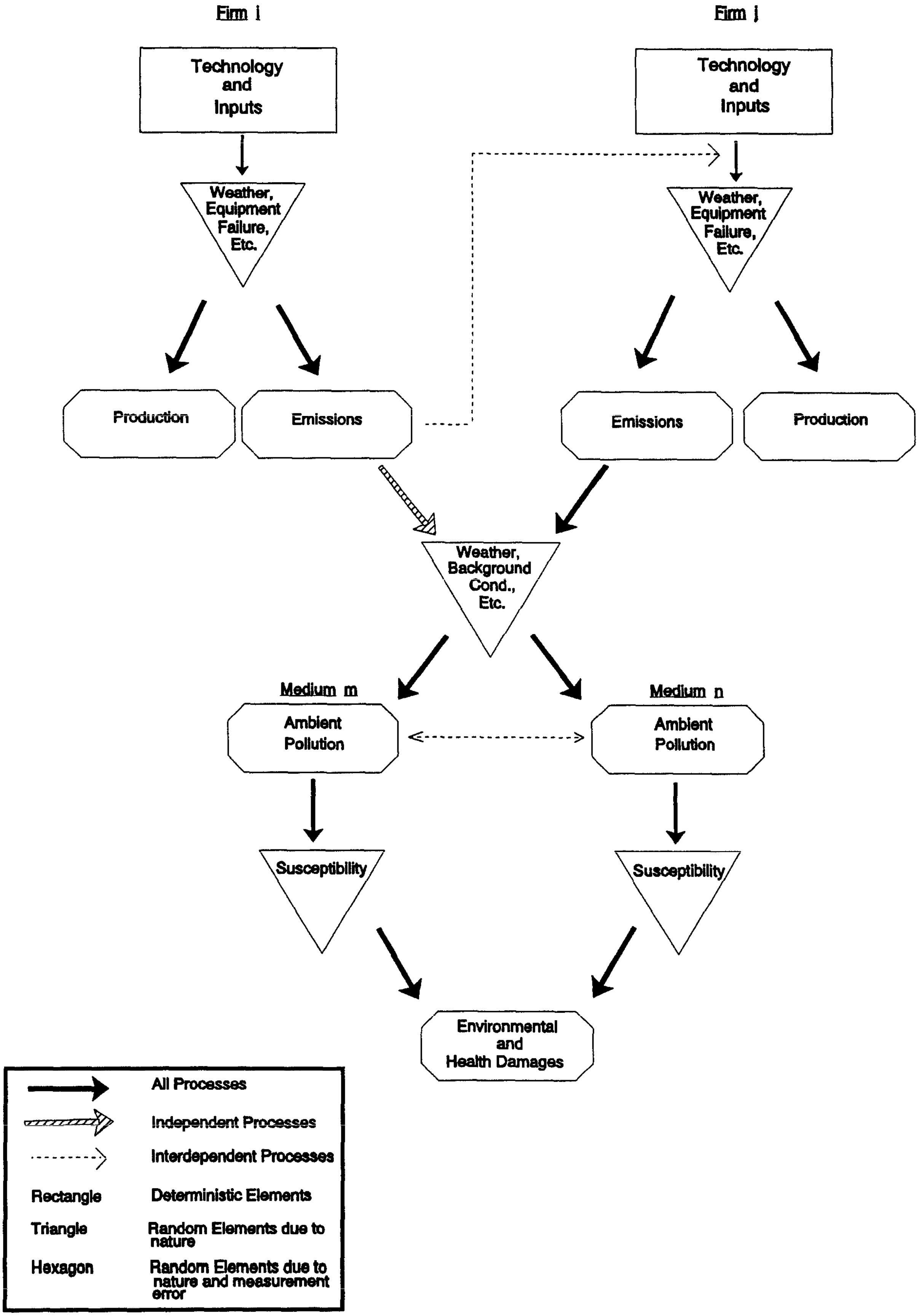
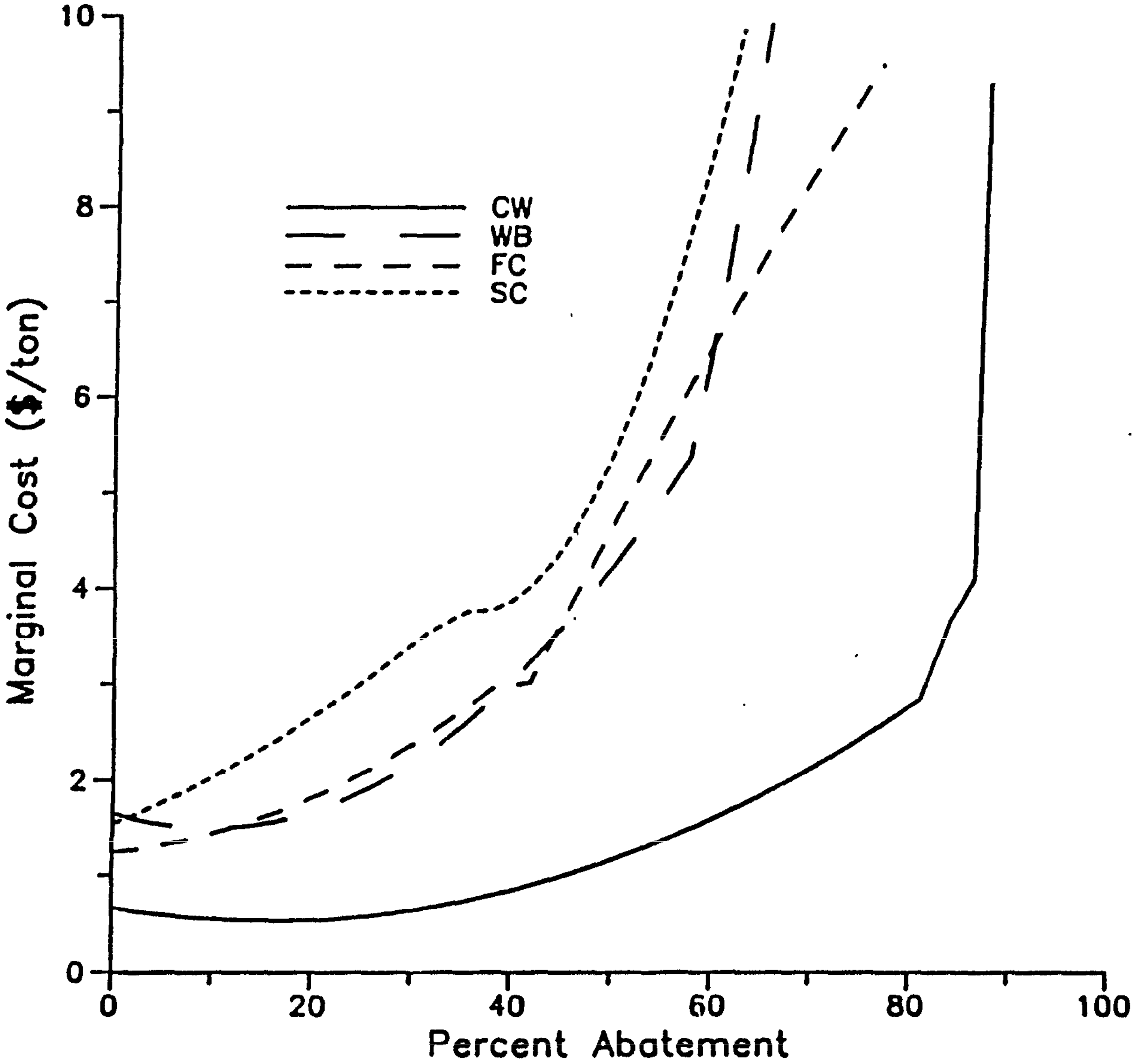


Figure 2. Illustrative Supply Curves for Sediment Abatement



Source: Braden *et al* (1989, p. 410)

Figure 3. Comparison of Input Tax Alone and Tax Plus Liability

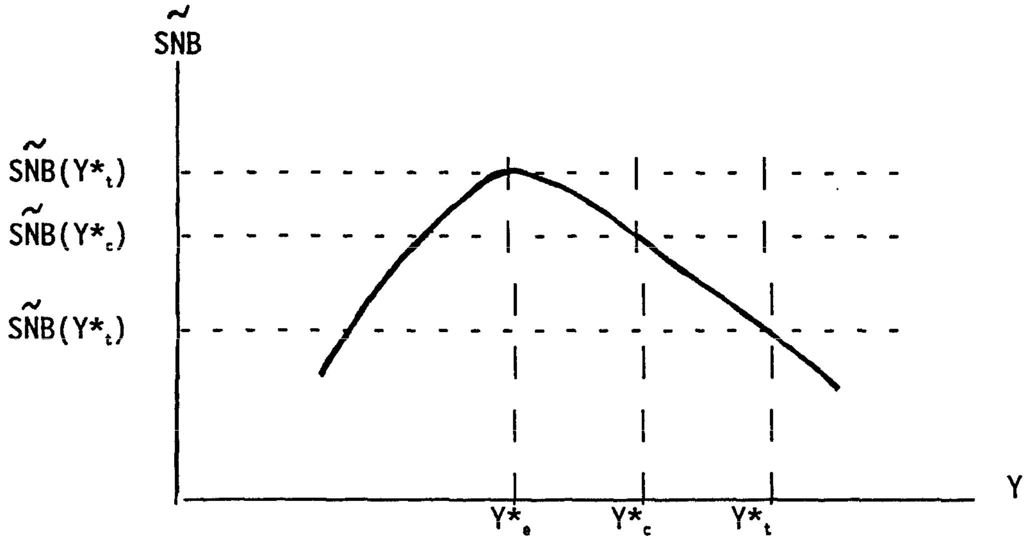


Figure 4. Comparison of Liability Alone and Liability Plus Tax
Case 1: $dY_L/dX > 0$

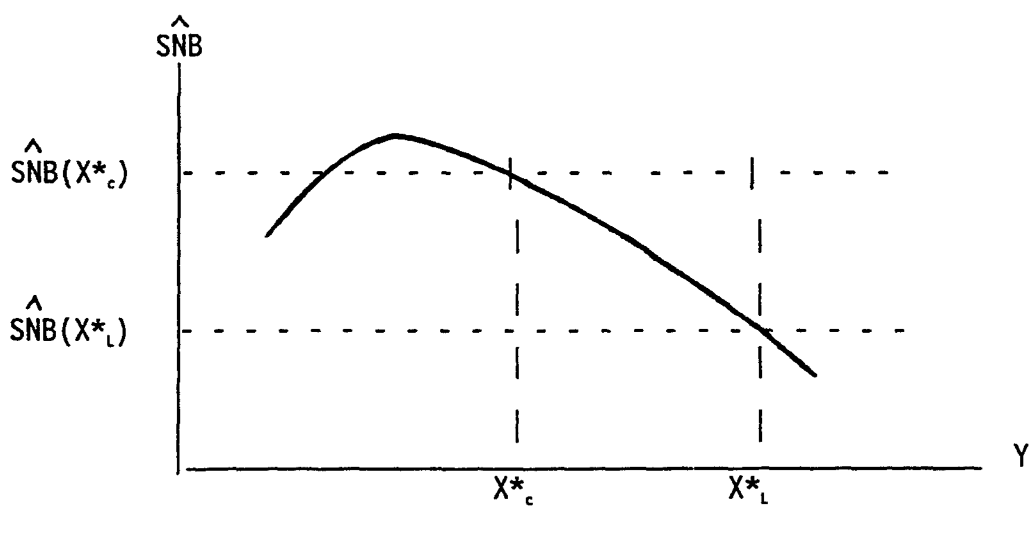


Figure 5. Comparison of Liability Alone and Liability Plus Tax
Case 2a: $dY_L/dX < 0$ and $\hat{SNB}_x > 0$ at X^*_L

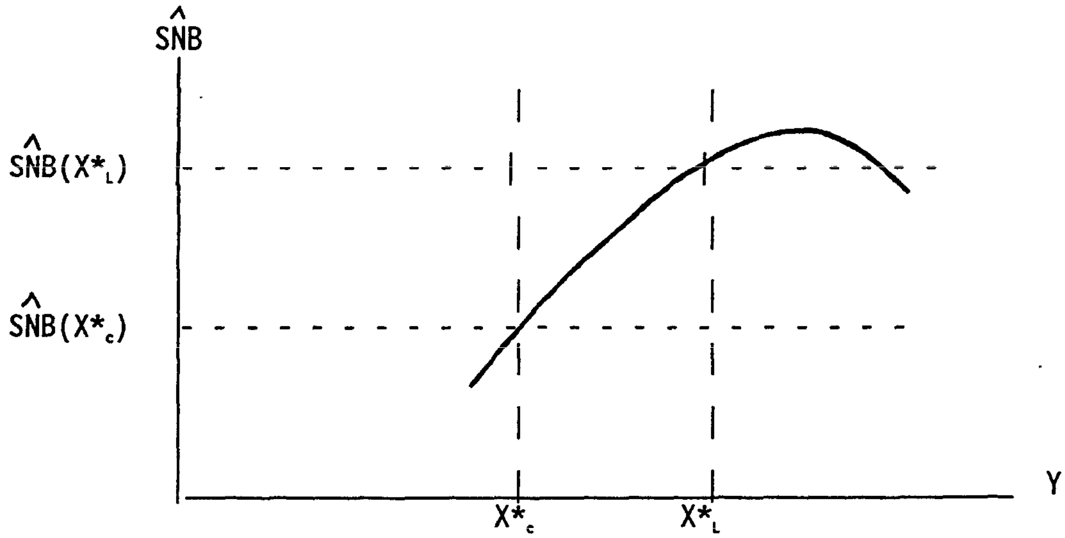
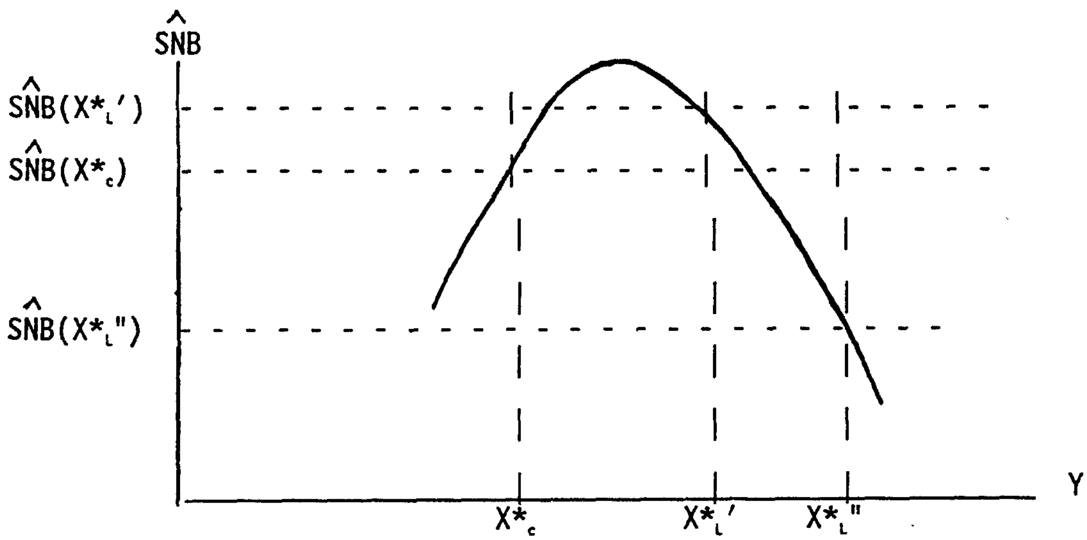


Figure 6. Comparison of Liability Alone and Liability Plus Tax
Case 2b: $dY_L/dX<0$ and $\hat{SNB}_x<0$ at X^*_L



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ENDNOTES

1. Examples of the second-best approach can readily be found in the extensive empirical literature on erosion and sediment control. Many studies in this literature evaluate the costs of public policies that clearly would institutionalize inefficiency, such as erosion regulations or mandated tillage practices(e.g., Lovejoy, Lee and Beasley 1985). Even more common in the economic literature is the consideration of policies one by one, rather than in combination. Examples come from our own work: Miltz, Braden, and Johnson (1988) compared the costs of reducing sedimentation via erosion taxes, erosion standards, and a spatially optimal plan. Segerson (1988) analyzed a tax on surface water ambient pollution due to agricultural emissions, and Segerson (1990b) considered liability remedies for ground water pollution by agricultural chemicals.

2. Agricultural nonpoint sources pollution exclude effluents from livestock confinement areas or spills or spills at chemical storage sites; these are generally regarded as point source problems and are regulated accordingly.

3. In addition to pollution processes, random variables also influence output levels. Concern for variation in output can affect the randomness of pollution outcomes. For example, a major concern for farmers is to hedge against the possibility of bad weather and bad crops. Crops, inputs, and farming techniques are chosen in part because of their potential for circumventing various risks--bad weather, bad prices, large fixed investments, and so on. Agricultural pollution stems in part from the choices that are made. Thus, government policies can influence polluting behavior indirectly by influencing the relative risks of different production systems (Kramer *et al.*, 1983; McSweeney and Kramer, 1986).

4. As shown by Weitzman (1974), with variable abatement costs and the need to select a policy instrument before costs are resolved, both incentives and regulations can be set to produce zero expected efficiency losses, but the costs of being wrong can differ greatly depending on the relative slopes of the damage and abatement cost curves.

5. With a flat supply curve and a steep demand curve, a regulatory instrument will tend to minimize the losses. On the other hand, with a steep supply curve and a flat demand curve, an incentive instrument will tend toward smaller losses.

6. While this paragraph makes particular reference to temporal variability, its conclusions also apply when a single policy is applied to a problem that varies through space.

7. Although standard instruments may be used, determining an optimal set of policies may be extremely difficult. Lichtenberg and Zilberman (1988) circumvent the problem by assuming a tractable functional form for damages More generally, however, Beavis and Walker (1983) analyze the case where the firms' discharges are independent and the goal is to limit the sum of realized emissions according to a constraint like (1). They show that, generally, the feasible set is nonconvex, so the first order conditions are not sufficient to ensure an optimum, and the regulatory or incentive measures identified through standard analyses will not necessarily be globally efficient. The search for efficiency would require detailed information and exhaustive analyses of alternatives. Convexity can also arise from interdependence among polluters. See, for example, the studies of sediment abatement by Bouzaher *et al.* (1990) and Braden *et al.* (1989).

8. For a somewhat outdated survey, see DeCoursey (1985).

9. Park and Shabman (1982) studied reductions of phosphorous. Reductions in sediment, phosphorus, and nitrogen were analyzed by Milon (1987). Braden *et al.* (1989) and Miltz *et al* (1988) considered the costs of reducing sedimentation. Braden *et al.* (1991) considered the costs of protecting fish habitat from sediment and pesticide pollution. Yet more numerous are studies of the costs of input restrictions (e.g., fertilizer or pesticide restrictions) without any linkage to environmental contamination or damages. These studies typically employ farm budgeting or linear programming techniques that involve fixed input combinations or tradeoffs. The Universal Soil Loss Equation (Wischmeier and Smith, 1978) is usually used to predict average annual erosion rates--there is no assumption that the rates could actually be observed in the field. Rarely are pesticide or fertilizer use rates transformed into measures of emissions.

10. Miles (1987) estimated the offsite value of erosion reduction to be about $1.00 per ton. Clark *et al.* (1985) estimated total offsite damages of $2.2 billion per annum for sediment in the U.S. If 10 percent to 30 percent of eroded soil accounts for these damages, the damages per ton would be between $0.75 and $2.30. Unfortunately, full-fledged demand (benefit) functions for abatement appear to be absent from the literature.

11. The supply curves in Figure 2 represent different assumptions about the relationship between erosion and sedimentation. The curve labelled CW reflects detailed simulation of the overland transport process including locations where deposition occurs. The curve labelled WB is based on a distance function--the closer source of the erosion, the higher the percentage of eroded soil that enters the water body. The curve labelled FC presumes that each field can be represented by a unique but fixed delivery ratio. This ratio does not change even though surrounding land uses change. Finally, the curve labelled SC is based on a single, fixed, average delivery ratio for the entire area. The curves CW, SC, and FC were calibrated to be directly comparable. The methodology behind WB cannot be directly compared to the others.

12. As evident in Figure 2, different methods of estimating the cost function may produce different curvatures and different conclusions about the type of instrument that will minimize realized errors. Here we have another information problem--limited understanding of pollution abatement options leads to the potential for substantial specification error in the estimation of abatement costs.

13. Russell, Harrington and Vaughan (1986), and more recently Harford (1990), show that monitoring costs and losses due to cheating can be diminished by a “state-dependent enforcement” scheme that makes the likelihood of monitoring and the regulatory standard conditional on a firm’s history of compliance. Those who have cheated and been caught would subsequently face more intensive monitoring and a tougher standard, and these threats help to induce compliance by former cheaters and potential cheaters alike.

14. See generally Besanko and Sappington (1987). Concerning environmental problems, Spulber (1988) analyzes an adverse selection problem while Shortle and Dunn (1986), Segerson (1988), and Xepapadeas (1991) consider a moral hazard problem. In Spulber's case, pollution abatement costs (which define the “type” of firm) are private information. An optimal policy entails abatement contracts which make polluters indifferent between falsifying their costs in an effort to gain a more lenient policy and truthful revelation of costs leading to a socially efficient standard. This is done by paying information rents. Such a policy is worthwhile only if the social benefit of less pollution is greater than the sum of the information rents plus the reduced economic surplus in the product markets to which the pollution is connected. In the other studies, the emissions or abatement efforts of individual polluters cannot be observed. Under these circumstances, an optimal policy involves a combination of fines and subsidies. One of the

instruments induces the optimal marginal incentives to abate while the other transfers income to counteract long-run distortions that can arise when the marginal incentives are determined by collective rather than individual emissions.

15. While this difficulty seems especially pronounced for nonpoint source pollution, indirect instruments also are used for point source applications in some instances when the costs of emissions monitoring would be prohibitive. This explains, for example, the use of design standards rather than emissions standards for various types of industrial fumes that are not easily captured in collection systems.

16. Holterman (1976) shows in a deterministic setting that efficient correction of an externality caused by a multiple input production process generally requires an instrument applied to each pollution-related input. Use of fewer instruments generally will lead to inefficient solutions. Nichols (1984) shows in a stochastic setting that input-based instruments will be more prone to inefficiency when the covariation between input use and external impacts is low; furthermore, the nature of the covariation can make incentive instruments superior to regulations (or *vice-versa)* in terms of efficiency.

17. This may be one area in which agricultural NPS is different from littering or other nonagricultural problems. For example, with waste disposal, cost-minimization promotes midnight dumping and other deceptive methods.

18. In contrast, the absence of measured emissions would undercut the use of a pure regulatory instrument applied to contamination. There would be no means of translating a contamination standard into abatement actions by individual polluters. However, such a translation could be achieved by combining an ambient standard with input or design restrictions, based on simulated predictions that the restrictions would achieve the standard. Alternatively, taxing contamination in excess of the standard, along lines suggested by Segerson (1988), could provide the necessary incentives.

19. Of course, the analysis could also be conducted for an abatement subsidy. A subsidy could achieve short-run efficiency but would boost agricultural profitability and encourage more of the polluting activity in the long run.

20. See Miceli and Segerson (1991) for an analysis of similar incentive problems in the context of liability for "joint torts", where the actions of several parties combine to determine a single level of expected damages.

21. Where the pollutant transport process causes physical interdependencies in the pollution transport process (Braden *et al.*, 1989), the efficient response of one party to a particular tax rate on ambient quality may depend on the actions of others . Under these circumstances, efficiency cannot be achieved through a fully decentralized price system.

22. The doctrine of joint and several liability offers a way around the need to determine the involvement of potential polluters. Miceli and Segerson (1991) discuss the efficient application of this doctrine to liability for environmental damages.

23. The use of multiple instruments to improve efficiency can be justified on other bases as well. For example, if there are multiple pollution-related inputs, then an input tax approach would require use of multiple input taxes (one for each input). Likewise, if there are multiple environmental media that are affected by a firm's activities, then an ambient-based approach would require use of multiple ambient-based taxes (or liability applied to multiple types of

damages). These examples do not, however, justify combined use of two different approaches such as the simultaneous use of input taxation and liability, which is the topic we consider in this section.

24. While we focus here on liability as a particular form of an ambient-based policy, we could alternatively have formulated the model in terms of ambient taxes. The results would be qualitatively similar, as long as the tax that would be paid by a given firm differs (with some probability) from the damages caused by that firm.

25. Note that the probability of being held liable for damages may differ across different media and different pollution types. For example, it may be easier to identify the source of a particular pesticide found in groundwater than the source of sedimentation in a given stream. For simplicity, we abstract from these issues here. Including them would complicate the notation without changing the basic qualitative results.

26. Of course, this is the same result that would be obtained if the base scenario were no liability and a tax were imposed on only one of the polluting inputs.

Working Draft

REGULATORY/ECONOMIC INSTRUMENTS FOR AGRICULTURAL POLLUTION: ACCOUNTING FOR INPUT SUBSTITUTION

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I. INTRODUCTION

In recent years, economists have explored the properties and relative merits of alternative instruments for the control of agricultural nonpoint-source pollution, such as design standards or incentives applied to farm inputs and performance-based standards or incentives applied to pollutant discharges. For the most part, such explorations have led to the conclusion that it is not practical at present to target either standards or incentives on agricultural nonpoint-source discharges directly, since their measurement is so difficult (e.g. Griffin and Bromley 1982; Dunn and Shortle 1988; Segerson 1988). As a result, much of the discussion regarding potential policies for agricultural pollution abatement has focused on restricting or providing negative incentives for the use of agricultural inputs and practices that yield pollution (such as nutrient fertilizers, pesticides, and erosive tillage systems).

A complicating factor in the design of instruments, however, is that a policy applied to one agricultural input can alter farm management practices and thus the utilization rates of other inputs. Such changes in input mixes have the potential to increase the release of pollutants that are different from the pollutant targeted by the instrument.

One example of potential input substitution involves tillage methods and their relationship to other inputs and practices used in agricultural production. Agricultural experts have long advocated conservation tillage as a best management practice (BMP) for soil consecration and the reduction of surface runoff of dissolved or sediment-bound pollutants from cropland. This BMP generally is defined as any type of tillage system that significantly lowers the erosion of soil by increasing the amount of plant residue (from the previous crop) that remains on the soil surface following planting. For example, where water erosion is the chief problem, consecration tillage is defined to be a system that maintains residue cover over at least thirty percent of the soil surface. The four main tillage types that satisfy this definition are no-till (which leaves the greatest amount of residue), mulch-till, ridge-till and strip-till (Soil Conservation Service 1989). The percentage of total acres farmed with conservation tillage in the United States has risen significantly in recent years, with the highest percentages now occurring in the Appalachian region (36% of total acres), the Northeast (33%), and the Corn Belt (33%) (Conservation Technology Information Center 1990).

Conservation tillage appears to be quite successful in reducing soil erosion and associated surface runoff of pollutants. However, experts have observed that the use of conservation tillage often is accompanied by increased use of nitrogen fertilizer (Crosson 1981), and studies have shown that under some soil conditions conservation tillage can lead to a significant increase in the infiltration of nitrogen into the subsurface (e.g. Alberts and Spomer 1985). In addition, the adoption of conservation tillage generally is associated with increased pesticide use (Crosson 1981; Epplin et al. 1982; Jolly et al. 1983; Duffy and Hanthorn 1984). Herbicide use tends to be higher under reduced tillage because conventional tillage serves to remove weeds In addition, reduced tillage creates moister soil conditions and more crop residue on the surface, both of which foster weed growth. (Residue increases the growth of weeds by isolating some of the applied herbicides.) The need for insecticides also tends to increase under conservation tillage because plant residue creates more favorable conditions for insects.

If an agricultural producer were able to increase sufficiently the use of integrated pest management (IPM) methods at the same time that he adopted conservation tillage, then it might be possible to reduce soil loss and associated surface runoff without significantly increasing the sheer volume of pesticides applied.(IPM decreases the necessity for pesticides through methods such as improved timing of planting and pesticide applications, use of resistant crop types, scouting, crop rotations, and biological pest control (National Research Council 1989).) However, the extent of this potential for IPM is not well understood at present. Furthermore, imperfect information regarding the private costs and benefits of IPM may lead to its suboptimal use. Thus while IPM may play an important role, evidence to date for specific crops clearly points to higher uses of pesticides under conservation tillage systems. Duffy and Hanthorn found this to be true for corn and soybeans in the major producing states, as did Epplin et al. for Southern Great Plains winter wheat.

As a consequence, conservation tillage may increase the potential for nitrogen and pesticides to escape from cropland and particularly to leach into groundwater. Conversely, it is conceivable that restrictions on herbicide use in areas that have exhibited increases in conservation tillage could force farmers to revert to conventional tillage methods, thereby exacerbating soil loss and surface runoff problems (Gianessi et al. 1988). Therefore a possible tension, or tradeoff, appears between the reduction of soil loss/surface runoff on the one hand and the infiltration of agricultural chemicals into groundwater on the other.

This paper presents an exploration of conceptual approaches to the problem of simultaneously managing multiple categories of agricultural pollution. First, the paper shows in simple fashion the way in which the "least-cost" allocation of pesticide abatement in an area may change if a link between pesticides and tillage is considered. (Though there also may be a relationship between tillage and nitrogen discharges, this paper focuses on pesticides.) Next, the paper describes a dynamic model that accounts for the possibly long-term damages that may result when pesticides leach into groundwater. The approach is useful in that it illustrates conceptually the source-specific characteristics that influence variability across areas and farmers in the desired degree of adoption of a BMP (in this case conservation tillage). Clearly, the information required to achieve an "optimal" tradeoff between different BMPs is far beyond reach. Therefore, the paper concludes by offering a few observations on possible ways to move closer to a least-cost approach to agricultural pollution control.

II. CONCEPTUAL APPROACHES

A. Allocating a Reduction in Pesticide Applications

Suppose that an environmental planning agency for a given area were to focus on the objective of reducing the area-wide rate of pesticide applications by a particular amount. This kind of objective would be similar to that established in 1987 for nutrients within the Chesapeake

Bay Agreement. Under that agreement, the U.S. Environmental Protection Agency and the states surrounding the Bay agreed to cut the loading of nutrients into Chesapeake Bay by forty percent by the year 2000. An objective of cutting area-wide pesticide applications, though, would be even simpler than the Chesapeake Bay nutrient objective in that it would not account for the relationship between application of pesticides at a given source and the loadings of pesticides to either surface water or groundwater.

Consideration of such a basic objective may be used to show how a linkage between pesticide application and tillage might affect the desired pattern of efforts to reduce the release of agricultural pollution in a particular area. First, suppose that no account were taken of the link between inputs. If the planning agency wished to achieve at least cost a reduction of a minimum of $\bar{p}$ in total pesticide use, then the problem would be similar in spirit to that which Krupnick (1989) illustrates for allocating the reduction of nutrient loadings to Chesapeake Bay, yet simpler due to the omission of discharge-loading coefficients:

$$\min_{p_j} \sum_j C_j(p_j) + \lambda_1(\bar{p} - \sum_j p_j) \qquad (1)$$

where: p_j = reduction in annual rate of pesticide application at Source j,

C_j = total annual private cost of reducing pesticide application at Source j,

$\bar{p}$ = desired annual reduction in area-wide pesticide application rates,

$C_j'(p_j) > 0,\ C_j''(p_j) > 0.$

The conditions representing the least-cost pattern of pesticide reduction, then, are:

$$C_j'(p_j) - \lambda_1 = 0, \quad \text{for all } j \tag{2}$$

$$\bar{p} - \Sigma p_j \leq 0, \ \lambda_1 \geq 0, \ \lambda_1(\bar{p} - \Sigma p_j) = 0 \tag{3}$$

Condition (2) shows that, if the planning agency wished to adopt an approach of restricting pesticide inputs, it could do so at least cost only if the input restrictions were made to vary so that the marginal cost of reducing pesticide use were equal for all sources. Alternatively, as Krupnick points out for nutrient reductions, a permit scheme could be established such that trading would take place until (2) was satisfied.

The socially efficient allocation of pesticide reduction in the area, however, would differ from that described by (2) if the total costs of reducing pesticide application included external costs not borne by the individual sources. That is, some producers might react to an instrument for pesticide reduction (be it an input restriction, tax, or tradable permits scheme) by substituting tillage operations for pesticides as a pest control method. A producer who does this will incur increased costs of labor and equipment necessary to conduct tillage operations. (Depending on the marginal effect of increased tillage, the producer also will incur a cost of foregone future productivity due to soil loss from farmland.) In addition, though, an increase in the surface runoff of pollutants will represent an external cost to society.

Ideally, the planning agency would develop a package of instruments that simultaneously takes into account the potential for surface runoff, the leaching of agricultural chemicals into the subsurface, and possible linkages between production practices that influence the magnitudes of both kinds of pollution. It is helpful, though, to consider initially the kind of problem the agency might face if it were to concentrate on the formulation of a policy for reducing one of the two kinds of pollution, say pesticide leaching. In this case, it might seek to attain the desired pesticide reduction objective while not causing changes in tillage practices that in turn would increase total surface runoff in the area above a predetermined acceptable increment.

This approach clearly would require some knowledge regarding the expected effect of a pesticide instrument on farm-level variables, including yield. In an empirical study, Gianessi et al. (1988) estimated the negative effect of a hypothetical local ban of a particular herbicide on farm production, and the associated consumer and producer welfare effects, for the Chesapeake Bay region. Because of the complexity of estimating input linkages, that study understandably did not attempt to examine the possible effect of such a ban on tillage practices and a consequent countervailing increase in yieldIt is useful for the present purpose of conceptually exploring the pesticide-tillage link to make a simplifying assumption about per-acre yield. Specifically, consider the problem the planning agency would face if it anticipated that producers would respond to an agency action by attempting to keep per-acre yields constant. (In what follows, the relaxation of this assumption would simply involve the introduction of an additional constant term denoting the expected percentage decrease in yield after accounting for anticipated changes in tillage.)

If producers wished to maintain per-acre yields constant, then in general a positive p_j at Source j would require that the producer substitute either or both of the following for the volume of pesticides applied: (1) an increase in tillage operations, or (2) IPM techniques that would allow the farmer to maintain yield while reducing pesticide applications and not increasing the intensity of tillage. Let:

$$\text{Level of Conservation Tillage Used at Source } j = T_j \tag{4}$$

where: $0 \leq T_j \leq \bar{T}$

$T_j = 0$ represents full conventional tillage,

$T_j - \bar{T}$ represents complete no-till farming,

and where intermediate values of T_j represent low-till systems, with increasing values for T_j reflecting higher "percent residue" levels. (Percent residue cover is an accepted way of comparing tillage methods.) It is assumed for simplicity that, for each source j, the method of tillage is homogeneous across all acres at Source j.

The change in T_j that is brought about by the pesticide instrument may be given as $t_j(p_j)$, where $\tau_j < 0$ represents a shift away from conservation tillage and toward conventional tillage, with $\tau_j'(p_j) < 0$. The value of $\tau_j'(p_j)$ is farm-specific and depends on the extent and nature of pest problems;the cost of managing pest problems with innovative IPM approaches; and factors that affect the desirability of conservation tillage, including soil productivity, perception of soil erosion (Gould et al. 1989), and operator tenure status (Hinman et al. 1983).

Define the change in the rate of surface runoff at Source j following a pesticide reduction policy as r_j, where $r_j > 0$ represents an increase in surface runoff. The change in runoff is a function of the change in tillage intensity:

$$r_j = \alpha_j \tau_j(p_j) \quad , \quad \alpha_j < 0 \tag{5}$$

where a_j denotes physical characteristics at Source j that influence the marginal effect of a change in tillage intensity on change in surface runoff. These characteristics would be those represented by the variables that appear in the Universal Soil Loss Equation, e.g. soil erodibility, precipitation, cropping, and farmland slope. Large absolute values for α_j would reflect conditions such as highly erodible soils, high rainfall, and steep land slopes. For simplicity a is assumed to be a constant, although in reality it might vary with the type of tillage employed.

Given these expected relationships, the planning agency could define its pesticide reduction problem as:

$$\underset{p_j}{\text{Min}} \sum_j C_j(p_j) + \lambda_1(\bar{p} - \sum_j p_j) + \lambda_2[\bar{r} - \sum_j(\alpha_j \tau_j(p_j))] \tag{6}$$

where: $\bar{r}$ = maximum area-wide increase in annual surface runoff that the planning agency wishes to allow,

and: $\lambda_1 \geq 0$, $\lambda_2 \leq 0$,

with the following necessary conditions:

$$C_j'(p_j) - \lambda_2 \alpha_j \tau_j'(p_j) = \lambda_1 , \quad \text{for all } j \tag{7}$$

$$(\bar{p} - \Sigma p_j) \leq 0 , \ \lambda_1 \geq 0 , \ \lambda_1(\bar{p} - \Sigma p_j) = 0 \tag{8}$$

$$[\bar{r} - \Sigma(\alpha_j \tau_j(p_j))] \geq 0 , \ \lambda_2 \leq 0 , \ \lambda_2[\bar{r} - \Sigma(\alpha_j \tau_j(p_j))] = 0 \tag{9}$$

Condition (7) shows that, as under the earlier agency problem, an interior solution requires that at every farm the marginal cost of reducing the application of pesticides should be set equal to the marginal benefit of doing so. Unlike in the earlier problem, the marginal cost of pesticide reduction now includes a term representing the environmental cost of an increase in surface runoff that is expected as producers respond by adjusting their tillage practices.

Condition (7) indicates the general way in which simultaneous incentives for pesticide reduction and conservation tillage should vary across areas so as to yield a least-cost solution to the agency's problem. Since tillage practices are observable by the agency (data exist already) and a tax/permit for pesticide use may be enforced, albeit imperfectly, at time of purchase, incentives targeted on these inputs would appear to be relatively practical from an enforcement standpoint.

B. Intertemporal Differences in Environmental Damages

1. A Simple Dynamic Model

While useful, the simple conceptual approaches above do ignore important aspects of the problem. First, they are based on the agency's objective of attaining at least cost those levels of pesticide application and soil erosion/surface runoff that it somehow has deemed to be acceptable. Under this kind of problem, the planner does not account for site-specific links between discharges at a given source and the environmental damages that are thereby generated. To take this into consideration, it would be necessary to develop some sort of damage function that would have as arguments the different kinds of agricultural discharges of interest.

Second, there may be interesting and important differences in the intertemporal patterns of damages generated by surface runoff and infiltration of pollution. For example, the persistence of pollutants can differ markedly depending on whether they leave farmland via surface runoff or leaching. Some toxic pesticides, for example Aldicarb, degrade rapidly in surface waters. Its degradation in groundwater, however, is much slower (Anderson et al. 1985). More generally, the degradation rates of many pollutants tend to be slower in groundwater due to the lack of sunlight, lower levels of oxygen, lower temperatures, and other physical, chemical, and biological conditions that are unfavorable to important degradation processes. Dynamic models that account for multiple pathways for pollution from a given source have been presented and simulated for an industrial waste stream (Eisworth 1988) and presented for agricultural pollution (Crutchfield and Brazee 1990). Krupnick (1989) uses a dynamic model to analyze damages from agricultural sources affect (which groundwater) and municipal treatment plants (which are assumed not to affect groundwater).

A useful way to incorporate the above elements is to consider the problem a planning agency would face if it wished to account for differences across sources in the link between production practices and environmental damages. Doing so provides insights on the way in which the desired degree of adoption of BMPs, such as conservation tillage and reduced use of persistent pesticides, may vary among geographic areas or producers. It is possible at the same time to incorporate dynamic factors. Consider, for example, the case in which the infiltration of pesticides into groundwater were to cause damages over a much longer period of time than pollutants carried from farmland in surface runoff. This is not to say that surface runoff cannot yield a long-lived flow of damages. However, it is fruitful conceptually to explore the extreme case where the environmental damages resulting at any point in time from the operation of a farm may be thought of as a function of: (1) the stock of existing pesticides that has built up in the subsurface due to pesticide applications in previous periods, and (2) the flow of pollutants that currently is escaping from the source via surface runoff. If the planning agency were interested in minimizing these environmental damages, then its instantaneous “utility function” for a given pollution source (farm) could be written as:

$$\text{Agency Function} = f(R,S)\ , \tag{10}$$

where: R = flow of surface runoff of pollutants from the farm,

S = stock of pesticides in groundwater resulting from applications on the farm,

$f(R,S) \leq 0,\ f_R < 0,\ f_{RR} < 0,\ f_S < 0,\ f_{SS} < 0,$

and where for simplicity the function is assumed to be additively separable so that $f_{RS} = f_{SR} = 0$. (With this kind of function, the agency's "utility", which is the negative of environmental damages, is always less than zero but may be increased by lowering surface runoff or the amount of pesticides in groundwater.)Next, let surface runoff at time t be a function of tillage method at time t:

$$\text{Surface Runoff} = R(T_t)\ ,\ R_T < 0, \tag{11}$$

where T is as defined above but no longer carries the subscript j because the level of analysis is now the individual source (farm). The sign and value of R_{TT} are dependent on physical conditions and presumably vary from farm to farm.

A portion of the pesticides applied to the farmland may be assumed to infiltrate into the saturated zone of the subsurface, and once there to undergo processes of natural decomposition into non-toxic substances. A general equation describing change over time in the stock of pesticides in the groundwater would be of the form:

$$\dot{S} = \beta Z_t - \alpha S_t \tag{12}$$

where: Z = rate of pesticide application,

α = mean rate of natural decomposition of pesticides in the subsurface, $\alpha > 0$,

β = proportion of total pesticides applied that migrate to groundwater, $0 < \beta < 1$,

and where β is dependent on factors such as soil permeability, rainfall, and depth to groundwater. The mean rate of decomposition, a, depends upon the characteristics of the chemicals applied and on physical conditions in the subsurface such as temperature, moisture, and chemical and hydrological characteristics.

Suppose that the agency is interested in encouraging the adoption of conservation tillage on farms in the area. How might the authority want the pattern of conservation tillage to vary spatially? The agency realizes that a shift toward conservation tillage may cause some producers to increase the intensity of pesticide use, but that the magnitude of such an effect would vary appreciably across farms. Given this, the agency might be interested in examining the way in which source-specific characteristics influence the desired level of conservation tillage at a given farm.

In order to do this most accurately in practice, it would be necessary to use a full model of agricultural production to estimate the response of all important variables, including the level of agricultural production, to an instrument that would require or encourage conservation tillage. Again, however, it is instructive to consider a much simpler model that focuses on the tension between minimizing damages from the surface runoff of pollutants and the infiltration of persistent chemicals. Assume therefore that in response to the agency's encouragement of conservation tillage, a given producer attempts to keep the rate of production q constant at $\bar{q}$. While this is a simplifying assumption, it may not be unreasonable for conceptual purposes, as some studies of input mixes under alternative tillage practices suggest that farmers who change tillage methods attempt to change the use of other inputs so as to keep per-acre yield at approximately the same level. (For example, Duffy and Hanthorn (1984) find differences in pesticide volumes and mixes, but no significant differences in per-acre yields, across different tillage practices for corn.)

A standard production model would have as inputs labor, capital, materials (such as pesticides and fertilizers) and land (e.g. number of acres and depth of soil). To examine the tillage-pesticide linkage, one may consider without loss of insight a partial production function such as:

$$q_t = q(Z_t, T_t) \qquad (13)$$

where $q_Z > 0$ and $q_T < 0$. If the producer is assumed to maintain $q(Z,T) = \bar{q}$, then Z may be expressed as a function of T, with $Z_T > 0$. The magnitude of Z_T will indicate several farm-specific characteristics, including the extent to which greater adoption of IPM would allow this particular producer to move toward no-till without applying a greater volume of pesticides. Though one might suspect that $Z_{TT} > 0$, its sign is not readily apparent and could vary across farms. An intertemporal model is made much more tractable by allowing Z_T to be a constant, and therefore let:

$$Z(T) = \delta T + \bar{Z} , \qquad (14)$$

where: $\delta > 0$, $\bar{Z} > 0$

and where $\bar{Z}$ represents the rate of pesticide application under conventional tillage.

Given such anticipated behavior of the producer, the planning agency might reasonably set as its goal the maximization of (10) net of the producer's expected costs of pesticide application and tillage operations, abstracting from other production costs such as those for seed and fertilizer. Such a planner's problem would be:

$$\max_{T_t} \int_0^{\infty} \{[f(R(T_t),S_t) - C_1(T_t) - C_2(Z(T_t)]e^{-rt}dt\} \tag{15}$$

$$\text{s.t.: } \dot{S}_t = \gamma T_t + \beta\bar{Z} - \alpha S_t \tag{16}$$

$$S_0 = \bar{S} \tag{17}$$

$$0 \le T_t \le \bar{T} \tag{18}$$

where: $C_1(T_t)$ = total private cost, at time t, of labor, fuel and repair, and machinery necessary for tillage operations; $C_{1T} < 0$; $C_{1TT} > 0$,

$C_2(Z(T_t))$ = total private cost, at time t, of labor, chemicals, fuel and repair, and machinery necessary for pesticide application; $C_{ZZ} > 0$; $C_{ZZZ} > 0$,

r = rate of discount,

$\gamma = \beta\delta > 0$.

The necessary conditions for this problem are:

$$f_T - C_{1T} - \delta C_{ZZ} + \gamma\lambda_1 \le 0,\ T \ge 0,\ T(f_T - C_{1T} - \delta C_{ZZ} + \gamma\lambda_1) = 0 \tag{19}$$

$$\dot{\lambda}_1 = \lambda_1(r + \alpha) - f_S \tag{20}$$

$$\dot{S} = \gamma T + \beta \bar{Z} - \alpha S \tag{21}$$

$$(\bar{T} - T) \geq 0, \ \lambda_2 \geq 0, \ \lambda_2(\bar{T} - T) = 0 \tag{22}$$

where for ease of notation time no longer explicitly appears as a subscript.

Condition (19) says that, for an interior solution, T should be set such that the marginal benefits of conservation tillage (reduced environmental damages from surface runoff plus reduced costs of labor, fuel and repair, and capital employed for tillage operations) equals the marginal costs of conservation tillage (an increased stock of pesticide in groundwater plus increased costs of labor, chemicals, fuel and repair, and machinery for the application of pesticide). Condition (20) shows that the optimal rate of change of the shadow price of the stock of pesticide in the groundwater depends on the instantaneous marginal damage caused by the pesticide stock, the rate of pesticide degradation, and the discount rate. Conditions (21) and (22) are the state equation and the Kuhn-Tucker conditions relating to the upper bound on T (complete no-till).

The dynamically optimal level of conservation tillage is given by the simultaneous solution of (21) and the steady-state condition for T. This condition is found by differentiating (19) with respect to time and substituting the result and (19) into (20), which after simplification gives:

$$\dot{T} = [(C_{1T} + \delta C_{ZZ} - f_T)(r + \alpha) - \gamma f_S]/[C_{1TT} + \delta^2 C_{ZZZ} - f_{TT}] \tag{23}$$

In steady state, then:

$$(C_{1T} + \delta C_{ZZ} - f_T)(r + \alpha) = \gamma f_S \quad (24)$$

Total differentiation of (24) shows that, as long as f_{TT} is either negative or, if positive, is less than $(C_{1TT} + \delta^2 C_{ZZZ})$, then the steady-state locus for T will slope downward in T-S space as shown in the phase diagram of Figure One. (The standard assumption is that f_{TT} is negative, which represents diminishing returns, in the form of reduced environmental damages from surface runoff, to conservation tillage.) As Figure One shows, a saddle point equilibrium exists for this problem.

One conceptual benefit of this model is that comparative statics analysis can show how changes in site-specific characteristics influence the optimal level of T. As an example, one may determine how the "desired" level of conservation tillage might vary from farm to farm according to variation in δ (the anticipated farm-specific link between tillage practice and the rate of pesticide application) and β (the proportion of applied pesticides that are expected to leach into groundwater). In this simple model, $y = \delta\beta$. The effect of a change in y is given by:

$$\partial T/\partial \gamma = -(\alpha f_S + \gamma f_{SS} T)/\{[-\alpha(r + \alpha)(C_{1TT} + \delta^2 C_{ZZZ} - f_{TT})] + \gamma^2 f_{SS}\} \quad (25)$$

Inspection shows that (25) is unambiguously negative, which is completely intuitive. In relation to δ, this means that for a farm at which one would expect to see a relatively high rate of substitution of pesticides

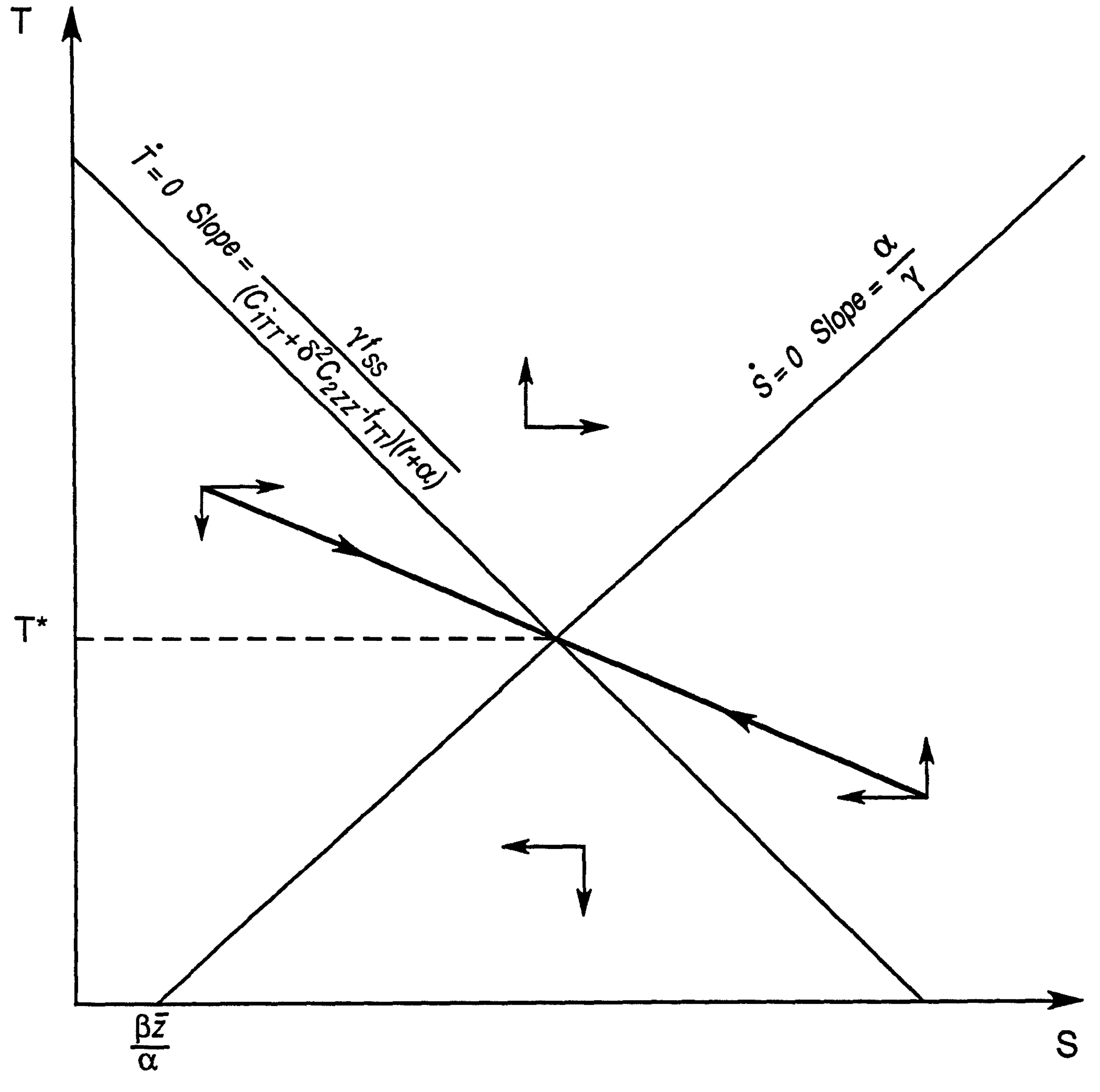
T
$\dot{T}=0$ Slope = $\frac{\gamma f_{SS}}{(C_{1TT}+\delta^2 C_{2ZZ}-f_{TT})(r+\alpha)}$
$\dot{S}=0$ Slope = $\frac{\alpha}{\gamma}$
T*
$\frac{\beta\bar{z}}{\alpha}$
S

Figure One

for tillage operations as a pest control method, the optimal level of conservation tillage would be relatively low, all else equal. With regard to β, this means that at a site exhibiting physical conditions that favor pesticide infiltration, the optimal level of T again will be relatively low, all else equal, as expected. The magnitude by which changes in δ and β would affect the desired degree of adoption of conservation tillage depends upon the private cost functions for pesticide application and tillage operations; the instantaneous damage function for surface runoff and pesticides in groundwater; the persistence of the pesticides; the rate of discount; and the values of y and T. Though the results are not shown here, one can use comparative statics analysis in this model to show the effects of changes in the other parameters on the desired level of conservation tillage.

2. Additional Considerations

The planner's problem shown above neglects an important consideration that may influence the pattern of adoption of conservation tillage that the agency wishes to encourage. One of the impacts of soil erosion is to reduce the agricultural productivity of land. That is, conservation tillage yields benefits to the agricultural producer in that it allows him to avoid the costs of foregone production that are imposed by soil erosion. The producer, however, may not take full account of this in his production decisions due to tenure status or misperceptions of erosion (Hinman et al. 1983; Gould et al. 1989). In addition, it would not be correct simply to fold this factor into the cost term $C_1(T)$, since a reduction in the intensity of tillage at time t yields benefits to the producer over all future time periods.

Instead, an appropriate agency problem that would capture this consideration would be:

$$\max_{T_t} \int_0^{\infty} \{[f(R(T_t),S_t) - C_1(T_t) - C_2(Z(T_t)) - C_3(E_t)]e^{-rt}dt\} \tag{26}$$

where: E_t = cumulative erosion, or soil loss, at time t;

$E(\bar{t}) = \int_0^t g(T_t)dt$, $g_T < 0$;

$g(T_t)$ = rate of erosion at time t;

$C_3(E_t)$ = total cost incurred at time t from lost agricultural production due to cumulative soil loss,

and where maximization of (26) would be subject to the same constraints as before plus an additional one:

$$\dot{E}_t = g(T_t) \tag{27}$$

With this objective function, the condition which maximizes the Hamiltonian with respect to T (for an interior solution) would differ from (19) only by the term denoting the addition to cumulative erosion:

$$f_T - C_{1T} - \delta C_{ZZ} + \gamma\lambda_1 + \lambda_3 g_T = 0 \tag{28}$$

where: λ_3 is a multiplier associated with the new constraint.

Such a framework could allow the agency to take account of the dynamic effects of lost productivity due to soil erosion. This would be most important for cases in which farmers do not perceive or take account of the full cost of foregone future productivity.

Lastly, uncertainty associated with parameter values clearly is a defining characteristic of the problems posed above. Sensitivity analysis therefore would be an important component of an attempt to simulate a dynamic model of tillage choice for a given site. Alternatively, uncertainty could be introduced explicitly by using a stochastic model of optimal control (e.g. Pindyck 1980; Kamien and Schwartz 1981).

C. Possibilities for Tailored Incentives

For any given source, there generally are large knowledge gaps regarding the kinds of parameters and functions featured in the conceptual approaches above. Furthermore, the expected values of and uncertainties associated with key parameters and functions vary appreciably across geographic regions and crop types. Policy clearly needs to account for such variation when addressing the tension between abatement practices for different pollution pathways. An important question, then, involves how this might be possible given constrained data on several counts and a limited understanding of pollution fate and transport processes, particularly in the subsurface.

Ideally, of course, planning agencies should like to implement a bundle of instruments that would bring about a least-cost movement to the "optimal" levels of different categories of pollution. In a less than ideal world, the agency might hope to develop instruments that would produce “charges and standards” results (Baumol and Oates 1975) for multiple pollution categories. These could consist, for example, of simultaneous instruments designed to achieve predetermined environmental quality changes through soil consecration (as a proxy for that class of surface runoff problems positively correlated with erosion) as well as chemical input use reductions.

In developing incentives for the control of multiple pollutants and pathways, a planning agency need not be concerned with tailoring the incentives according to producers’ private costs and benefits of abatement, since producers will account for those factors in deciding how to respond to incentives. The key lies in accounting for variation across areas, producers and crops in the external effects of BMPs that policies encourage. If incentive (fee, subsidy or permit) schemes were implemented simultaneously for both pesticide use and tillage practice, then the total costs of reducing environmental damage would be lowered by varying the incentives spatially according to area-specific parameters and functions such as α, β, δ, R(T), and f(R,S). (In an expanded model allowing for the "containment" of pollution in addition to the reduction of discharges (Braden et al. 1989), the total costs of damage reduction could be lowered even further.)

Given limited information, a practical approach to the development of tailored incentives might involve identifying a small number of specific ranges into which key parameter values may fall. Then, an agency could proceed to build a taxonomy that identifies, by crop and spatial location, the expected place that each key parameter is thought to occupy in the classification. For some key parameters, the information necessary to characterize their expected ranges is already available. For example, good information by location is available for the variables of the Universal Soil Loss Equation. This means that though the precise specification of runoff as a function of tillage may be difficult, one certainly can draw general conclusions about the way in which the relationship varies across areas and producers. Information on other factors, of course, is less available. The effect of different agricultural production practices on water quality, for example, is not well understood at present. Research has been underway on these factors, and plans for new studies currently are being developed by the U.S. Geological Survey and the U.S. Department of Agriculture (Burkart et al. 1990). This kind of research should enhance the base of knowledge regarding spatial variation in factors such as δ, α and β and the marginal damages associated with the discharge of pollutants from cropland.

Even in the presence of incomplete information on factors such as pollutant fate and transport in groundwater and the effect of farming practices on various discharges, it is possible with current knowledge to make general distinctions among areas. This is demonstrated quite well by Crutchfield et al. (1991) through their classification of the vulnerability of groundwater to pesticide and nitrate leaching from cotton production in different states. Their work estimates the percentages of cotton cropland in the major producing states that fall into four distinct categories of vulnerability to pesticide leaching, running from "most vulnerable" to "little or no likelihood" of leaching. The same is done for nitrates, with three categories corresponding to high, moderate, and low vulnerability. These kinds of estimates could provide useful input to tailor the magnitudes of incentives for BMP adoption according to the agricultural production and environmental characteristics of different geographic areas.

III. SUMMARY

A policy designed to decrease the use of an agricultural input that causes pollution can lead farmers to alter their management practices and thus the overall input mix. This may lead in turn to an increase in the discharge of pollutants different from those targeted by the policy. One example involves substitution between tillage operations and the application of pesticides. A policy to decrease pesticide pollution by lowering the rate of pesticide application may cause an increase in erosive tillage practices and thus soil loss and associated surface runoff. Alternatively, a policy designed to increase conservation tillage may yield higher damages from pesticides.

This paper has explored conceptual approaches to the management of agricultural nonpoint-source pollution that take account of substitution between tillage operations and pesticides. Under the simple objective of reducing the total discharge of pesticides in an area by a given amount, the least-cost allocation of abatement changes if input substitution is accounted for and a constraint on surface runoff is imposed. The allocation of pesticide reductions would change according to farm-specific factors such as soil erodibility and productivity, rainfall, cropping, farmland slope, severity of pest problems, the potential and cost of "integrated pest management," farmer perception of soil erosion, and farm operator tenure status. This paper also presented a dynamic model of a planning agency’s choice of tillage at the farm level that accounts for the potentially long-term damages that may result when pesticides leach into groundwater. The approach illustrates the tradeoff between reducing surface runoff of pollution and the leaching of pesticides, and shows how cross-farm variability in key parameters would influence the desired degree of adoption of conservation tillage.

In the conceptual approaches of this paper, it is assumed that farmers respond to an instrument targeted at an agricultural input by altering other inputs so as to maintain constant per-acre yields. Though some data on agricultural practices suggest that farmers may attempt this, a more realistic approach would relax this assumption to allow for a decline in yield. Useful further work also would include explicit treatment of uncertainty regarding parameters; consideration of a range of pesticides with varying effectiveness, toxicity and persistence, among which farmers may choose; and an exploration of the impact of integrated pest management techniques on the extent of input substitution.

Since tillage practices are observable and disincentives for pesticide use may be applied at the time of purchase, instruments targeted directly at these inputs are relatively practical. Empirical application of conceptual approaches for even a few agricultural sites could help to determine a ranking of priorities for fine tuning a package of instruments according to local agricultural and environmental factors. For a given set of multiple environmental objectives, one could determine how relatively sensitive an efficient solution is to variation in different parameters and functions (e.g. β vs. $R(T)$), and thus identify the most critical "driving" characteristics upon which the tailoring of instruments might be based. This in turn would increase the efficiency of pollution control efforts by directing future research toward those parameters that are found to be most important.

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(preliminary version)

NON POINT SOURCE POLLUTION CONTROL , INFORMATION ASYMMETRY, AND THE CHOICE OF TIME PROFILE FOR ENVIRONMENTAL FEES

by

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1. INTRODUCTION

Water quality management specialists have long ago emphasized the practical difficulties of relying on "conventional" end-of-pipe control techniques when dealing with non-point pollution (NPP) problems, and, hence, the need to prevent pollutant loads as far as possible.

The application of such a "preventive approach" through effective and possibly efficient, regulatory schemes, may, however, involve number of problems, which to a large extent arise from the difficulty, and sometimes the technical impossibility, of monitoring non-point emissions at source. This may be due either to the mode of conveyance of pollutant flows, or to the intermittent nature of emissions or to the fact that pollutants originate over a broad area [Vigon, 1985]. The relative role played by each of these factors in preventing the monitoring of emissions on a continuous and widespread basis may vary according to the specific pollutant at hand.

Due to the difficulty of applying *sic et simpliciter* emission-based policy instruments when dealing with NPP, attempts have been made to find alternatives, with respect to actual emissions at source, as a basis for establishing regulatory schemes. In this respect, basically two recommendations can be found in economic literature dealing with NPP control.

The first consists of selecting incentives defined with respect to ambient pollutant levels, According to its proponents, the proposal would have the attractive property of allowing regulatory bodies to rely on "...an incentive mechanism based on the observable variable (ambient pollutant levels) to induce certain unobservable actions [pollutant abatement at source]" [Segerson, 1988, P.89].

The second policy strategy, which may be termed an "indirect approach", suggests regulatory bodies should grant political legitimacy to NPP mathematical models which make predictions about either emissions at source or ambient pollutant levels, and hence define appropriate incentives accordingly. As the proponents state, ". . . while such models will never provide [..] a perfect substitute for accurate monitoring of actual flows, they can serve as an important tool for diminishing the uncertainty about nonpoint loadings . . . . furthermore, predictions obtained from such models offer an alternative to actual flows as a basis for the application of policy instruments [Shortle-Dunn, 1986, p.668].

The two above-mentioned approaches will be briefly reviewed and commented on in section 2. While there are no a *priori* decisive arguments in favor of one or the other, we suggest that the "indirect" one should be preferred whenever there are no indications that the suspected polluters possess better information about the "technology" of pollutant abatement. Or, more generally, whenever it is believed that the cost of acquiring information about the implications of productive decisions in terms of ambient pollutant levels are prohibitively high for the private economic agents involved.

It is worthwhile stressing, however, that granting political legitimacy to a NPP mathematical model does not constitute, *per* *se,* a panacea. In fact, from a regulatory point of view, it simply implies that the availability of adequate information about the parameters needed to feed the model, rather than monitoring of actual emissions, becomes the key issue when establishing policy instruments.

Such parameters are usually represented by productive decisions taken by suspected polluters, such as the use of (potentially) polluting inputs (eg. nitrogen fertilizers) and the physical characteristics of the production site (eg. soil water retention capacity). In addition, models conceived as tools for providing not only estimates of potential pollutant flows but also pollutant transport rates, typically require information about the hydrological structure of the watershed surrounding the water body which is thought to receive a fraction of estimated emissions at field level.

Acquiring such information may not constitute a serious problem for regulatory bodies operating in Countries or regions with long-standing traditions of land classification and where management practices are monitored on a continuous and widespread basis. However, in Countries which do not have such traditions the application of the "indirect approach" may be problematic and lead to unsatisfactory results, unless regulatory bodies are either prepared to invest resources in collecting the required information directly, or to "extract" it from suspected polluters through appropriately defined incentive mechanisms.

The paper concentrates on the application of the policy strategy which has been referred to as the "indirect approach". In particular, we try to make a step forward with respect to a previous work where we attempted to provide, through a static, discrete "adverse-selection" framework, a broad characterization of NPP incentive schemes when suspected polluters possess private information about their production site's physical characteristics (Dosi and Moretto 1990). Here, on the other hand, we shall assume that at the time when the regulatory scheme is designed, the suspected polluters do not possess such private information. However, the assumption that the relevant physical characteristics of the production site ("soil quality") do not vary over time, will be relaxed,

In fact, it appears more realistic to assume that a number of physical characteristics affecting the extent of pollutant loads (as well as, often, suspected polluters' productive performance) may vary over time. Changes may either occur because of "exogenous shocks" or because of actions taken by firms or both. In the paper we concentrate on non-monitorable actions undertaken by firms ("maintenance decisions"), but it will be assumed that the "maintenance technology", known by both parties, is affected by a certain degree of uncertainty.

Furthermore, we account for the possibility that, even if the social planner (hereafter the agency or the (p)rincipal) perceives the existence of detrimental externalities due to the presence of unregulated non-point pollutant sources, he might consider the opportunity of delaying the introduction of "environmental fees". Such fees will be assumed to take the form of an increase in the market price of the variable input which is believed to contribute to emissions, and, hence, water contaminations with an intensiveness which depends on the production site's physical characteristics. Moreover it will be assumed that suspected polluters grant enough credibility to the agency's announcement concerning the time profile of environmental charges.

The implications of such a delay as well as its optimal choice characterization constitute the paper's central issues.

Section 3 explores the implications of the announcement of alternative delays upon management decisions adopted by the suspected polluter(s) -hereafter the firm or the (a)gent)- and on environmental damage. In the same section we also analyze the action of uncertainty about future realizations of the soil quality index, with regard to the firm's maintenance pattern and its "market value", as well as with regard to the consequent (expected) environmental damage.

Section 4, on the other hand, will be devoted to the optimal choice characterization of the time profile for environmental fees, by assuming the perspective of a utilitarian agency which, over the entire planning horizon, takes care of (expected) estimated social environmental damages as well as of the firm's welfare, and receives a utility from tax collection. The optimal choice will be derived by looking for a "perfect equilibrium" within a two player game in which the firm chooses management practices after observing the agency's time profile decision. The equilibrium is then obtained by working backward: the agency foresees that the firm will react optimally to whatever time delay for environmental fees is announced. That is, the agency should solve the firm's optimization problem before taking its own decision.

2.THE RATIONALE FOR AN “INDIRECT” NPP CONTROL POLICY APPROACH

2.1 Before turning to the paper’s main issues, let us briefly review the two general NPP control policy strategies mentioned in the previous section.

As a point of reference, let us start by assuming that the firm’s fixed-capital output per unit (say bushels of corn per acre) is given by the following production function:

[1] $$Q = Q(\theta, x)$$

where x represents a (potentially) polluting variable input (eg. chemical fertilizer) and θ represents an index for fixed-capital quality (eg. soil water retention capacity).

Let us distinguish between (unobservable) pollutant emissions at source, R, and pollutant levels actually found in a given water body, P. Assume, for the time being, that R depends only on x, $R = R(X)$, P is linked to R by a one-to-one relationship, $P = P(R)$, and social damages associated with P are evaluated according to $D = D(P)$.

Given such relationships, a variety of policy options are, at least theoretically, open to the agency. Such options range from restrictions on the permissible x level the firms will be allowed to choose, to incentives defined over observable ambient pollutant levels. Placing an incentive over P (or R, since it may be easily inferred from observing P), however, appears to be the most attractive option from an "administrative" point of view. Moreover, if the firm possesses better information about the "technology" of pollutant abatement, placing an incentive over P (or R) should induce the selection of cost-minimizing pollutant abatement strategies.

However, if this condition is not met, or, more generally, if the firm does not possess a priori information about R(.) and P(.), the agency should transmit all the relevant information, but in this case we do not see any reason why the agency should not "convey" such information either through appropriately defined incentives over x or through mandatory measures.

Whether or not defining standards in terms of permissible x will provide results allocatively equivalent to management practice incentives will depend on whether or not the agency possesses adequate information about θ, a state variable which, for the time being, we assume only affects the firm'smarketable output. Setting aside transaction costs, if the parties share the same information about θ, the two policy instruments would lead to allocatively similar results, since an optimal incentive scheme over x should induce a profit maximizing firm to choose the same variable input level which the agency would choose as a management practice standard. However, if the firm has better information about θ when the agency implements the policy, such equivalence breaks down, and the pricing mechanism is preferable to standards since it has the advantage of relieving the principal from the problems posed by the definition of a standard on x in conditions of uncertainty regarding θ.

To summarize, two considerations follow from the stylized “technical” relationships which have been assumed up to now.

Firstly, if the firms do not possess a *priori* information either on R(.) or P(.), whilst the agency does, a regulatory approach directly defined over the production decisions affecting pollutant flows appears more appropriate than defining incentives over actual ambient pollutant levels (or on R), since the theoretical advantages of assigning the firms the role of choosing the best abatement strategy can not be exploited.

Secondly, if the firms possess private information about θ, management practice incentives appear to be allocatively superior to management practice standards. According to Shortle and Dunn [1986], in conditions of imperfect monitorability of the firms' “typology”, such incentives would not only outperform all the alternative policy instruments but, at least in the special case of a single (suspected) polluter, they would lead to achievement of a first-best solution: a result which, however, if our interpretation is correct, crucially depends on the assumption that θ does not enter R(.) (or P(.)).

2.2 Let us now modify the technical relationships p(.) and R(.) so as to make them a little closer with “reality”.

First of all, let us specify P(.) in a way which formalizes the NPP attribute consisting of the difficulty ofinferring, emissions at source, without errors, from observable ambient pollutant levels. This can be done either by incorporating in the argument a random variable representing the imperfect knowledge about pollutant transport mechanisms:

$$P = P(R,\tau)$$

or by allowing for the existence of multiple sources:

$$P = P(R_1, \ldots\ldots\ldots, R_n)$$

or both:

$$P = P((R_1,\tau), \ldots\ldots\ldots, (R_n,\tau))$$

How does recognition of the existence of a more complex relationship between ambient pollutant levels and each source's emissions affect the conclusions previously drawn when assuming to deal with a one-to-one relationship?

A first consideration should be made with regard to the need to distinguish between emissions at source and ambient pollutant levels. If the agency wishes to improve social welfare, and not to reduce emissions as such, accounting for a more complex and articulated relationship between R and P clearly emphasizes the need to implement policy instruments which take ambient pollutant levels rather than emissions at source as their point of reference. In this respect, we entirely share the view expressed by Segerson [1988], according to whom economic incentives concentrating on the latter tend to ignore "...the important distinction between "discharges" and the resulting pollutant levels which determine damages" [p.87].

However, whether or not this objective may be better accomplished by relying on incentives defined on actual ambient levels or through management practice incentives defined according to NPP mathematical models (providing predictions about P) will depend, again, on how plausible we believe is the assumption that private agents possess, or may easily acquire, information about the implications of their management practices in terms of ambient pollutant level. It is clear, however, that the more complex the P(.) relationship is, the less this assumption appears to be plausible.

Further complications arise if we assume that also the relationship between the firm's productive decisions and emissions at source is more complex that the one assumed up to now. For example, R might be influenced not only by use of the potentially polluting input as such, but also by the physical characteristics of the site in which x is used, i.e.

$$R = R(x,\theta)$$

Incorporating θ in the argument of R(.) appears, in fact, more consistent with the technical literature dealing with NPP, which suggests that climatic, pedological and topographic parameters may play an important role in determining the extent of pollutant quantities potentially affecting surface and underground water quality.

Unfortunately, once a more complex P(.) function is combined with a more complex relationship between R and management practices, it becomes even more difficult to share the optimism expressed, for example, by Segerson [1988] that, "... since firms are in a better position to determine the abatement strategy that will be most effective for them", the selection of incentives defined over actual ambient pollutant levels would ensure that "...any given level of abatement is achieved at the lowest possible cost" [Segerson, 1988, P.86].

This optimism may be justified if we assume we are dealing with relatively "simple" and, at least in certain Countries, long experienced phenomena such as erosion, but is less convincing when dealing with inherently more complex, and less “perceptible”, pollution phenomena such as nitrate and pesticide leaching.

Moreover, it should be pointed out that the difficulties met by individual polluters in identifying the relationship between their productive decisions and the level of pollution appearing in (often distant) water bodies, and, then, in conjecturing their own responsibility, increase with the number of sources: in fact, even with respect to the same pollutant, within the same watershed point and non-point sources may be contemporaneously present, and, among the latter, differentiated urban, industrial and agricultural polluting activities. This is, for example, the case of the watershed surrounding the Venice lagoon, a water body in which worrying phenomena of algae-bloom have occurred repeatedly in recent years: this densely populated area of about 180,000 hectares; located in Northern Italy, presents an extraordinary mix of industrial and agricultural activities which, together with the urban sector, emit quantities of nitrate and phosphorus estimated, respectively at approx. 9,000 and 1,300 tons per year [Regione del Veneto, 1989].

In such conditions, it appears, in our view, more suitable to rely on what has been termed the"indirect approach", and hence convey information to firms about NPP mathematical model-based ambient pollutant predictions through appropriate incentives directly placed on management practices.

2.3 The paper thus focuses on this policy approach, and concentrates on some issues related with its adoption.

We assume that at the time when the regulatory framework is decided, firms do not hold private information about the production site's physical parameter(s), $\boldsymbol{\theta}$, entering the NPP mathematical model which has been granted with political legitimacy. It will be assumed, however, that the soil quality index, characterizing each production unit, may vary over time. In fact, if the agency is assumed to take a sufficiently long planning horizon, accounting for the possibility that $\boldsymbol{\theta}$ will evolve with respect to its initial status, appears more realistic than assuming it remains invariable.

Let us take the example of "nitrate emissions" from cultivated soils, a phenomenon often considered, at least in EEC Countries, as one of the most relevant problems among NPP. Technical literature suggests that "discharges" are undoubtedly positively correlated with fertilizer use; however, leaching of available nitrates may significantly increase due to high rates of water movement through the soil. In turn, high water movement may be due to (more or less) unpredictable heavy rainfall conditions which cannot be prevented. However, farmers may contribute to reducing very high water movement, for example, by takingactions designedto maintain the soil's organic content, since soils rich in terms of organic matter have, generally, relatively high water retention capacity, and are therefore liable to experience lower losses of available nitrates [OECD, 1986]. Again, however, the performance of such actions in terms of maintaining (or not depleting, or increasing) organic content, and, then, the consequences in terms of final ambient pollutant levels, may be affected by a certain degree of uncertainty) depending on a number of (more or less) unforeseeable factors [Regione Veneto, 1990].

On the grounds of this example, two considerations are in order.

Firstly, it would be inappropriate to disregard the possibility of a variation over time of the soil quality index θ which enters the NPP mathematical model upon which ambient pollutant predictions are based, and, hence, on which regulatory schemes are defined. To go back to our example, whether this index refers to "soil water balance" or to "organic content of the soil", θ is unlikely not to vary over time.

Secondly, θ may vary both because of exogenous "shocks" and actions undertaken by firms. It follows that, even if we concentrate on the latter, it appears convenient to assume that "maintenance decisions" are undertaken in conditions of uncertainty about future realizations of thesoil quality index which the firm (the agency) wishes to alter in order to improve its profits (social welfare).

An attempt to deal formally with such issues is made in the paper. It is assumed that the firms' "maintenance expenditures" are not monitorable by the agency, but that both parties share the same information about maintenance technology and uncertainty with regard to future realizations of θ. The assumption of identical information about the structure of the maintenance technology function appears not too unrealistic, since agencies themselves may provide the firms with all the relevant information they possess about the possible performance of maintenance actions.

As far as the "form" of uncertainty is concerned, θ is assumed to move randomly in continuous time according to the following stochastic differential equation:

$$[2] \qquad d\theta_t = \left[f(\theta_t, m_t) - \delta \right]\theta_t dt + \sigma\, \theta_t dz_t$$

where f(.) stands for the effect of maintenance expenditure, m, δ is a constant soil quality "depreciation" rate, and dz is the increment of a Wiener process, or Brownian motion, with zero mean and unit variance (i.e. $dz_t = \varepsilon_t \sqrt{dt}$, while ε is a serially uncorrelated and normally distributed random variable)[1].

Equation [2] implies that the future realizations of θ are uncertain with a variance which grows linearly with the time horizon, Thus, although information is obtained over time, future soil quality status is always uncertain to the firm.

We assume that:

$f_m > 0$ (i.e. maintenance expenditure has a positive influence on θ),

$f_{mm} < 0$ (i.e. this influence diminishes as m increases),

$f_\theta < 0$ (i.e. for a given amount of maintenance expenditure the improvement of "low quality soil" is greater than for "high quality soil"),

$f(0,\theta) = 0$,

$\delta > 0$ (i.e., if the firm decides not to spend money on maintenance, the expected value of θ deteriorates at the constant exponential rate δ).

It is assumed that the firm wishes to maximize its "market value" , i.e. its discounted (expected) cash flows over the planning horizon $[0 , \infty)$. According to [1] and [2], and setting output price equal to one, the firm's objective function in the absence of public intervention is described by:

$$E_0\left\{ \int_0^{\infty} \left[Q(x_t,\theta_t) - \omega\, x_t - m_t \right] e^{-rt} dt \right\}$$

where ω indicates the input market price faced by a competitive, representative firm and r is a constant discount rate,

As, according to our hypothesis, the agency might announce the decision to delay the introduction of “environmental fees”, the firm’s objective function with regulation becomes:

$$[3] \quad V(\theta_0;T) = E_0\left\{ \int_0^{T} \left[Q(x_t,\theta_t) - \omega\, x_t - m_t \right] e^{-rt} dt + \int_T^{\infty} \left[Q(x_t,\theta_t) - \omega\, x_t - D(P(\theta_t))x_t - m_t \right] e^{-rt} dt \right\}$$

where T represents the time lag “granted” to firms before introducing a tax which is assumed to take the form of an increase in the price of the variable input x. The amount of this increase will depend on the social damage, D(P), attributed to ambient pollutant levels, evaluated according to a mathematical model, $P(\theta)$, which is assumed to provide variable input (x) predictions per unit.

3. THE EFFECT OF TIME PROFILE ON FIRM'S MANAGEMENT PRACTICES AND THE ROLE OF UNCERTAINTY

3.1 Let us assume the production function [1] has the following properties:

[4]
$$\begin{cases} Q_x > 0 \,, & \text{for } x \le \bar{x} \\ Q_x < 0 \,, & \text{for } x > \bar{x} \\ Q_{xx} \le 0 \,,\; Q_\theta > 0 \,,\; \bar{x}_\theta > 0 \\ Q(0,\theta) = Q(x.0) = 0 \,,\; Q_x(0,\theta) = \infty \end{cases}$$

Since x may be freely adjusted, the firm's optimal variable input level can be derived from the usual first order condition:

[5]
$$\begin{aligned} &Q_x - \omega = 0 && \text{for } 0 \le t < T \\ &Q_x - \omega - D = 0 && \text{for } t \ge T \end{aligned}$$

If we set, for simplicity, $\omega = 0$, according to [4] and [5] the optimal input level will be[2]:

[6]
$$\begin{aligned} &x^{*}_{(a)t} = \bar{x}(\theta_t) && \text{for } 0 \le t < T \\ &x^{**}_{(a)t} = Q_x^{-1}[D(\theta_t)] && \text{for } t \ge T \end{aligned}$$

where, for a given θ_t, $x^{*}_{(a)t} \ge x^{**}_{(a)t}$.

BY substituting [6] in [3] the latter reduces to:

$$[7] \qquad V(\theta_0;T) = E_0\left\{ \int_0^T \left[Q(\bar{x}(\theta_t), \theta_t) - m_t\right] e^{-rt}dt + \right.$$

$$\left. + \int_T^\infty \left[Q[Q_x^{-1}(D(\theta_t)),\theta_t] - m_t - D(\theta_t)Q_x^{-1}(D(\theta_t))\right] e^{-rt}dt \right\}$$

To keep the problem mathematically tractable, we shall assume that:

$$[8] \qquad \begin{aligned} Q(x,\theta) &= h(\theta)x^{\alpha} &&, \quad 0 < \alpha < 1 \\ h(\theta) &= \theta^{\nu} &&, \quad \nu > 0 \\ D(P(\theta)) &= \theta^{-\beta} &&, \quad \beta > 0 \\ \bar{x}(\theta) &= \theta^{\psi} &&, \quad \psi > 0 \\ f(\theta,m) &= m^{\xi}\,\theta^{-\gamma} &&, \quad 0 < \xi < 1, \quad \gamma > 0 \end{aligned}$$

Then the firm's maximization problem becomes:

$$[9] \qquad \max_m V(\theta_0;T) \equiv E_0\left\{ \int_0^T \left[\theta_t^{\varphi} - m_t\right] e^{-rt}dt + \int_T^\infty \left[C_{(a)}\theta_t^{\phi} - m_t\right] e^{-rt}dt \right\}$$

where $C_{(a)} = (1-\alpha)(\alpha)^{\alpha/(1-\alpha)} < 1$

$$\phi = \frac{1}{1-\alpha}\nu + \frac{\alpha}{1-\alpha}\beta$$

$$\varphi = \nu + \alpha\psi$$

The maximization is subject to equation [2], the constraint $m \geq 0$, and θ_0 is given. Moreover we assume that the sample path of $\{z,\}$ contains all the information relevant to the firm's problem, and $E_0\{\cdot\}$ denotes conditional expectation taken, at time zero, over the distribution of $\{z_t\}$ and $\{\theta_t\}$ processes. While the former is exogenous to the firm's problem, the latter is determined endogenously by the optimal maintenance pattern.

According to [9], the firm's maximization problem can be set as a two-stage optimal control problem, where the integral assumes different forms in each stage. In the second stage the firm maximizes its expected discounted cash flows, defined as the difference between "operational profits" and environmental fees. Then, in the first stage, the firm will maximize its discounted operational profits, with the constraint that at time T the firm's market value will coincide with the (discounted) value calculated in the second stage.

Let us then solve the optimal control problem at II-stage, formally expressed as follows:

$$[10] \quad \max_{m} e^{-rT} V^{II}(\theta_T) \equiv e^{-rT} E_T\left\{ \int_T^{\infty} \left(C_{(a)}\theta_t^{\phi} - m_t\right) e^{-r(t-T)} dt \right\}$$

The maximization is subject to equation [2], m ≥ 0, and θ_T given. If the firm's maximum market value at the II-stage is differentiable, then $V^{II}(.)$ has to be a solution of the following dynamic programming equation:

$$[11] \quad rV^{II} = \max_{m} \left[\left(C_{(a)}\theta_t^{\phi} - m_t\right) + V_{\theta}^{II} \left(m_t^{\xi}\, \theta_t^{-\gamma} - \delta\right)\theta_t + \right.$$
$$\left. + \frac{1}{2}\sigma^2\, \theta_t^2\, V_{\theta\theta}^{II} \right]$$

where V_{θ}^{II} and $V_{\theta\theta}^{II}$ are derivatives of $V^{II}(.)$ with respect to 8.

Equation [11] is the Hamilton-Jacobi-Bellman equation of the stochastic version of the optimal control theory. By differentiating the r.h.s of [11] with respect to m_t , we obtain:

$$[12] \qquad m_t = \left(\xi \, V^{II}_{\theta} \, \theta_t^{1-\gamma} \right)^{1/(1-\xi)} \qquad \text{for} \quad t \geq T$$

which represents the first order condition for optimality of the firm's maintenance pattern.

Equations [11] and [12] together can be expressed as a non-linear second order differential equation of parabolic type in V^{II}. As pointed out, for example, by Freedman (1964), Merton (1975), such a differential equation, in general, can not be solved explicitly. However, if some restrictions on the coefficients of the production, damage, and maintenance technology function are imposed, it may be possible to find a solution in a closed analytical form. In particular, if we set(3):

$$\xi = \frac{1}{2} \quad , \quad \gamma = \frac{1}{2}\phi$$

the solution for the firm's market value is (see appendix A):

$$[13] \qquad V^{II}(\theta_t) = M^{II}\theta_t^{\phi} \qquad \text{for } t \geq T$$

where $$M^{II} = \frac{B - \sqrt{B^2 - 4AC_{(a)}}}{2A}$$

$$A = \frac{1}{4}\phi^2$$

$$B = r + \phi\delta - \frac{1}{2}\phi(\phi-1)\sigma^2 \quad > 0$$

$$C_{(a)} = (1-\alpha)(\alpha)^{\alpha/(1-\alpha)}$$

According to [13], the firm's optimal market value at the

II-stage is an increasing function of the state variable θ with elasticity equal to ϕ, which, in turn, depends on the production function's parameters α and ν, and on β, the elasticity of the social damage function with respect to θ.

From [13] the optimal maintenance policy can be derived:

[14] $$m_t = \frac{1}{4}\phi^2 \left[M^{II}\right]^2 \theta_t^{\phi} \qquad \text{for} \quad t \geq T$$

which implies the stochastic differential equation [2] reduces to:

[15] $$d\theta_t = \left[\frac{1}{2}\phi\, M^{II} - \delta\right]\theta_t dt + \sigma\, \theta_t dz_t \qquad \text{for} \quad t \geq T$$

From [14], the optimal maintenance policy at the II-stage depends on the current realization of the state variable θ. Since, in turn, θ is described by a stochastic process, also m will be a stochastic process. In other words, the firm can not decide on maintenance expenditure before looking at the "performance" of θ achieved through past maintenance decisions.

We can now examine maximization of the firm's market value at the I-stage, on condition that it coincides, at time T, with the discounted value described by [13], which, in this optimal two-stage control problem, takes on the sense of "scrape-value". Formally the firm's expected discounted profit or loss of at time zero, described by [9], becomes:

[16] $$\max_{m} V(\theta_0;T) \equiv E_0\left\{ \int_0^T \left(\theta_t^{\varphi} - m_t\right) e^{-rt} dt + e^{-rT} M^{II} \theta_T^{\phi} \right\}$$

Now the firm faces the problem of maximizing [16] by choosing a maintenance expenditure policy in the interval [0,T) with a terminal constraint at time T. In our case, for $\theta \geq 0$ and a generic $0 \leq t < T$, [16] can be rewritten as:

$$[16] \quad \max_{m} V(\theta_t, t; T) \equiv e^{rt} E_t \left\{ \int_t^T \left(\theta_s^{\varphi} - m_s \right) e^{-rs} ds + e^{-rT} M^{II} \theta_T^{\phi} \right\}$$

Hence $V(\theta_t, t; T)$ is the maximum profit at time t if the soil quality index at that time is θ multiplied by e^{rt},

Again the maximization is subject to equation [2], $m \geq 0$, and θ_t given. The procedure for solving [16'] is the same as that used for the II-stage. In other words, if the market value function of the firm V is differentiable, then $V(\theta_t, t; T)$ has to be a solution of the following dynamic programming equation:

$$[17] \quad -V_t + rV = \max_{m} \left[\left(\theta_t^{\varphi} - m_t \right) + V_\theta \left(m_t^{\xi} \theta_t^{-\gamma} - \delta \right) \theta_t + \right.$$

$$\left. + \frac{1}{2} \sigma^2 \theta_t^2 V_{\theta\theta} \right] \quad , \text{ for } 0 \leq t \leq T$$

with the following constraints:

$$V(\theta_T; T) = M^{II} \theta_T^{\phi}$$

$$V(0; t) = 0$$

where V_t is the partial derivative of V(.) with respect to t.

By differentiating the r.h.s of [17] with respect to m, we get:

[18] $$m_t = \left(\xi\, V_\theta\, \theta_t^{1-\gamma}\right)^{1/(1-\xi)} \qquad \text{for} \quad 0 \le t < T$$

Again, to obtain a solution in a close analytical form, we need to impose some restrictions on the “technical” coefficients. In particular, if we set(4):

$$\xi = \frac{1}{2}, \qquad \gamma = \frac{1}{2}\varphi \left(\equiv \frac{1}{2}\phi\right)$$

the solution for the firm’s market value at I-stage is (see appendix 1):

[19] $$V(\theta_t, t;T) = M(t;T)\,\theta_t^{\varphi} \equiv M(t;T)\,\theta_t^{\phi} \qquad , \text{ for} \quad 0 \le t < T$$

where:

$$M(t;T) = \frac{M_2^I - M_1^I \left(\dfrac{M^{II} - M_2^I}{M^{II} - M_1^I}\right) \exp\left(\sqrt{B^2 - 4A}\right)(T-t)}{1 - \left(\dfrac{M^{II} - M_2^I}{M^{II} - M_1^I}\right) \exp\left(\sqrt{B^2 - 4A}\right)(T-t)}$$

$$M_1^I = \frac{B - \sqrt{B^2 - 4A}}{2A} \quad , \quad M_2^I = \frac{B + \sqrt{B^2 - 4A}}{2A}$$

It is easy to show that $M(T;T) = M^{II}$ and the following limits hold:

$$\lim_{T\to\infty} M(t;T) = M_1^I$$

$$\lim_{T\to 0} M(t;T) = M^{II}$$

with $M_1^I \le M^{II}$ as indicated in fig.1. In addition, since

$\frac{M^{II} - M_2^I}{M^{II} - M_1^I} > 0$ (see appendix B) if the introduction of environmental fees were postponed forever, the firm's maximum expected value would be reached at t = 0, and would decrease over time. On the other hand, if no delay were conceded, the firm's maximum value would be that obtained in the II-stage solution.

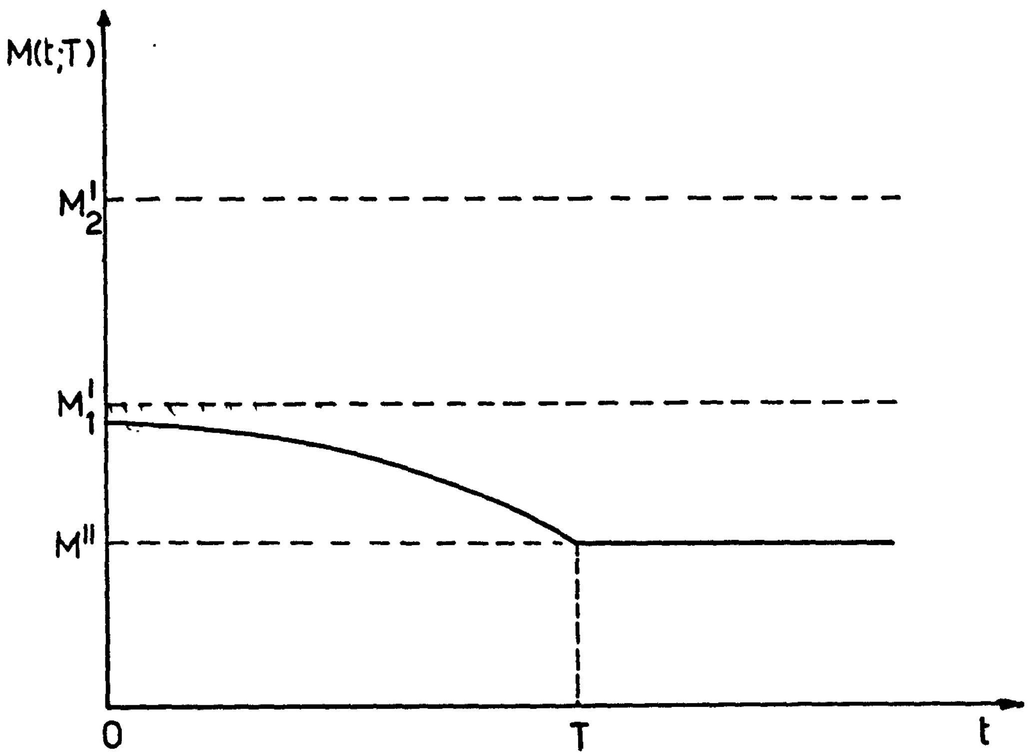

fig.1

The optimal maintenance expenditure pattern during the period preceding introduction of the environmental fees can be derived from [19]:

$$[20] \quad m_t = \frac{1}{4} \phi^2 \left[M(t;T) \right]^2 \theta_t^{\phi} \qquad \text{for } 0 \leq t < T$$

while the stochastic differential equation [2] reduces to:

[21] $$d\theta_t = \left[\frac{1}{2}\phi\, M(t;T) - \delta \right]\theta_t dt + \sigma\, \theta_t dz_t \ , \quad \text{for} \quad 0 \le t < T$$

Again, m_t appears to be a stochastic process, and the firm can not decide on the optimal maintenance expenditure in advance before looking at the current realization of the state variable θ.

3.2 On the basis of the results obtained above, let us now explore: (i) the relationship between the time profile announced by the agency, T, and the expected maintenance expenditure pattern; (ii) the relationship between T and expected total damage; (iii) the action, at equal T, of uncertainty with regard to future realizations of soil quality parameter θ on the (expected) maintenance expenditure, the firm's market value and environmental damage.

3.2.1 Since m is a stochastic process, the expected value of its rate of variation can be derived by applying the **Itô's** Lemma to [20]:

[21] $$\frac{1}{dt} E_t\left(\frac{dm_t}{m_t}\right) =$$

$$= \begin{cases} 2\,\dfrac{\dot{M}(t;T)}{M(t;T)} + \phi\left[\dfrac{1}{2}\phi\, M(t;T) - \delta\right] + \dfrac{1}{2}\phi(\phi-1)\sigma^2 \ , & 0 \le t < T \\ \phi\left[\dfrac{1}{2}\phi\, M^{II} - \delta\right] + \dfrac{1}{2}\phi(\phi-1)\sigma^2 \equiv r - \sqrt{B^2 - 4AC_{(a)}} \ , & t \ge T \end{cases}$$

By solving the differential equation [21] taking the expectation at time zero, we obtain: